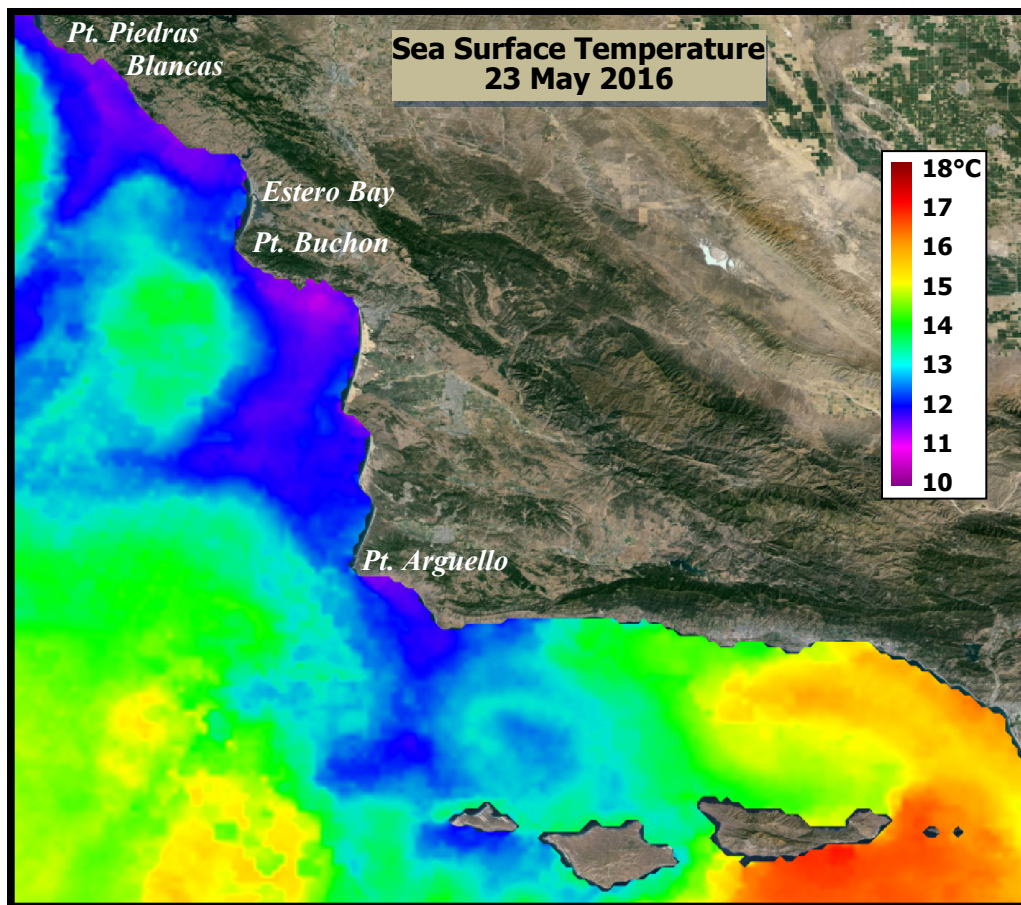


**City of Morro Bay and
Cayucos Sanitary District**

OFFSHORE MONITORING AND REPORTING PROGRAM

2016 ANNUAL REPORT



Marine Research Specialists

4744 Telephone Rd., Suite 3-315
Ventura, California 93003

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**Report to the
City of Morro Bay and
Cayucos Sanitary District**

**955 Shasta Avenue
Morro Bay, California 93442
(805) 772-6272**

**OFFSHORE MONITORING
AND REPORTING PROGRAM**

2016 ANNUAL REPORT

**Prepared by
Douglas A. Coats**

**Submitted by
Marine Research Specialists**

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March 2017

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marine research specialists

4744 Telephone Rd., Suite 3-315 • Ventura, CA 93003 • 805-644-1180

John Gunderlock
Wastewater & Collection Systems Supervisor
City of Morro Bay/Cayucos Sanitary District
955 Shasta Avenue
Morro Bay, CA 93442

31 March 2017

Reference: 2016 Annual Monitoring Report

Dear Mr. Gunderlock:

Enclosed is the referenced report. It documents the continued effectiveness of the treatment process, the absence of marine impacts, and compliance with the discharge limitations and reporting requirements specified in the NPDES discharge permit.

Please contact the undersigned if you have any questions regarding this report.

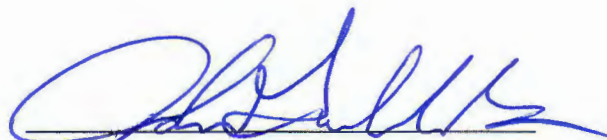
Sincerely,

 **MA**RINE **RE**SEARCH **SP**ECIALISTS
Vice President

2017.03.31 15:36:00 -07'00'

Douglas A Coats
Project Manager

I certify under penalty of law that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to assure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, the information submitted is, to the best of my knowledge and belief, true, accurate and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations.

A handwritten signature in blue ink, appearing to read 'John Gunderlock', written over a horizontal line.

Mr. John Gunderlock
Wastewater/Collections System Supervisor
City of Morro Bay/Cayucos CSD Wastewater Treatment Plant

Date: 4/3/17

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LIST OF ACRONYMS AND DEFINITIONS

ACOE	U.S. Army Corps of Engineers
Anomaly	An anomaly is a deviation or departure from ambient (mean) conditions in one of the six measured seawater properties: salinity, temperature, density, dissolved oxygen, alkalinity, or transmissivity. The amplitude of the anomaly is quantified as the difference between the measured seawater property at a given station and the mean of that property measured at all other stations at the same depth level or distance above the bottom.
Anthropogenic	Changes to the environment induced by human activities
APCD	San Luis Obispo County Air Pollution Control District
ASP	Amnesic Shellfish Poisoning
BCCE	Brown and Caldwell Consulting Engineers
BEHP	Bis 2-Ethyl Hexyl Phthalate is a compound that leaches out of plastics and is ubiquitous in the environment.
BIP	A Balanced Indigenous Population is as an ecological community that exhibits characteristics similar to those of nearby, healthy communities existing under comparable but unpolluted environmental conditions.
BBL	A benthic boundary layer is often synonymous with a bottom nepheloid layer (<i>see below</i> BNL) and is associated with increased turbulence and mixing caused by the decrease in current flow immediately above the seafloor that is required to meet the zero-velocity boundary condition at the seafloor.
BNL	The bottom nepheloid layer is a common phenomenon on continental shelves where increased turbidity near the seafloor is caused by the presence of naturally occurring particulates formed from light-weight flocs of detritus. Oscillatory bottom currents generated by passing surface gravity waves easily suspend these flocs.
BMPs	Best Management Practices
BOD₅	Five-day biochemical oxygen demand
BTEX	Benzene, toluene, ethyl benzene, and xylene
Cal/OSHA	California Department of Industrial Relations, Division of Occupational Safety and Health
CAM-17	CAM is the California Administrative Manual and CAM-17 refers to the list of 17 heavy metals identified in the California Code of Regulations (CCR 2003), Title 22, Chapter 11: Identification and Listing of Hazardous Waste.
CCLEAN	<u>Central Coast Long-term Environmental Assessment Network</u>
CEQA	California Environmental Quality Act
CCR	California Code of Regulations (CCR 2003)
CDFG	California Department of Fish and Game
CIWQS	<u>California Integrated Water Quality System Project</u>

LIST OF ACRONYMS AND DEFINITIONS

(Continued)

CDP	Coastal Development Permit
CMAR	Construction Manager at Risk: A project delivery method wherein the Construction Manager (CM) is required to deliver a project within a guaranteed maximum price
Coordinate Datum	All coordinates cited in this report are referenced to the WGS84 datum
COP	The California Ocean Plan (SWRCB 2005), which is regularly revised but the 2005 edition was in force when the current NPDES discharge permit was issued to MBCSD.
CSWP	Cayucos Sustainable Water Project
CTD	A CTD is an instrument package that continuously records conductivity, temperature, and depth, in addition to a number of other physical oceanographic parameters in the water column.
CV	A coefficient of variation is used to compare the relative amounts of variation in populations having different means. It is the standard deviation expressed as a percentage of the mean.
CWSRF	Clean Water State Revolving Fund (CWSRF) that includes a Water Recycling Funding Program administered by the SWRCB Division of Financial Assistance
DDT	Dichloro Diphenyl Trichloroethane, a chlorinated pesticide
DEHP	Di(2-ethylhexyl) phthalate synonymous with Bis(2-ethylhexyl) phthalate
dEIR	Draft Environmental Impact Report
DHS	California Department of Health Services
DMR	Discharge Monitoring Report
DMR-QA	Discharge Monitoring Report Quality Assurance (DMR-QA) Study administered by the United States Environmental Protection Agency
DO	Dissolved Oxygen
EIR	Environmental Impact Report
ELAP	California Department of Health Services, Environmental Laboratory Accreditation Program
ENSO	El Niño–Southern Oscillation
EQ	Exceptional Quality is a standard for compost that meets the Federal and State requirements for beneficial reuse as a soil amendment in the local community.
ERL	Effects Range Low is a sediment toxicity concentration below which adverse biological effects are not expected to occur.
ERM	Effects Range Median is a sediment toxicity concentration above which adverse biological effects are expected to occur.
ESA	Endangered Species Act

LIST OF ACRONYMS AND DEFINITIONS

(Continued)

FOG	Fats, Oils, and Greases
GPM	Gallons per Minute
HAB	Harmful Algal Bloom
Hazwopper	Hazardous Waste Operations and Emergency Response
HCl	Hydrochloric Acid
HCH	Isomers of hexachlorocyclohexane including alpha, beta, gamma (lindane), and delta
I&I	Inflow and Infiltration
Isopycnal	A surface of constant seawater density
JPA	The Joint Powers Agreement between the City of Morro Bay and the Cayucos Sanitary District (MBCSD) outlines the contractual agreement between the two agencies for the operation of the WWTP
Leachate	A solution formed by leaching, especially a solution containing contaminants picked up through the leaching of soil
LC₅₀	Lethal 50% Concentration is a toxicity endpoint where 50% of the specimens will die after a 96-hour exposure period.
LOE	Line of Evidence used to evaluate whether a receiving-water measurement complied with permit limits
MANOVA	Multivariate Analysis of Variance
MBAS	Monterey Bay Analytical Services
MBCSD	The City of Morro Bay and Cayucos Sanitary District
MCC	A Motor Control Center manages the speed and torque of electric motors connected to the Center
MTBE	Methyl Tertiary Butyl Ether is an additive to gasoline that boosts octane to reduce air emissions but has contaminated groundwater in many regions after accidentally leaking from underground storage tanks.
MDL	Method Detection Limit is the lowest concentration that can be reported under ideal conditions where the sample contains only the compound of interest with a concentration in an optimal calibration range and in a medium that does not interfere with the performance of the analytical instrument.
MGD	Million Gallons per Day
mg/Kg	Milligrams per Kilogram = $\mu\text{g/g}$ dry weight = parts per million
mg/L	Milligrams per Liter = aqueous parts per million
Mixed liquor	A mixture of activated sludge, wastewater, and oxygen where organic sewage constituents are biologically assimilated

LIST OF ACRONYMS AND DEFINITIONS

(Continued)

ML	The Minimum Level is the method-specific minimum concentration of a substance that can be quantitatively measured in a sample given the current analytical performance used by most certified laboratories within California, as specified in the 2005 Ocean Plan.
MLLW	Mean Lower Low Water
MMRP	Major Maintenance and Repair Plan
MMS	Minerals Management Service
MPN	Most Probable Number
MRS	Marine Research Specialists
MT	Metric Ton = 1,000 kg
MWH	Montgomery, Watson, Harza is the engineering firm selected for the design of the MBCSD treatment-plant upgrade.
NEP	National Estuary Program
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No Observable Effect Concentration
NORM	Naturally Occurring Radioactive Material
NPDES	National Pollutant Discharge Elimination System
NTU	Nephelometric Turbidity Units
O&G	Oil and Grease
O&M	Operations and Maintenance
OIT	Operator-in-Training
OSHA	Occupational Safety and Health Administration
PAH	Polynuclear Aromatic Hydrocarbons
PCA	Principal Component Analysis is a multivariate technique that arranges samples along an axis based on species composition. This arrangement is conducted independently along a number of dimensions (usually 2 or 3) that approximate some pattern of response of the intertidal community to underlying environmental gradients. PCA condenses the complex species-abundance database to a few factors responsible for observed variability within the intertidal community structure, while retaining ecologically meaningful biological information.
PCB	Polychlorinated Biphenyl
PCi/L	Pico-Curies per liter is a measure of aqueous radioactivity.

LIST OF ACRONYMS AND DEFINITIONS

(Continued)

Perturbation	A perturbation is a small but detectable variation (anomaly) in one or more seawater properties caused by the presence of dilute effluent within the receiving waters. Individual Perturbations identified in this report are designated P1 through P12.
PG&E	Pacific Gas and Electric
POTW	Publicly Owned Treatment Works
Power (1-β)	The probability correctly rejecting the null hypothesis depending on the significance criterion, the variability of the sample results, and the size of the impact
ppm	Parts per million = mg/L in solution , or mg/Kg = μg/g dry weight
PQL	Practical Quantification Limit is the lowest concentration that can be measured with statistical reliability given the sample size and analytical method.
PSDWF	Peak seasonal dry-weather flow
PSP	Paralytic Shellfish Poisoning
<i>p</i>-value	Low <i>p</i>-values (<0.05) indicate a high degree of confidence that the computed quantity is statistically significant.
PWWF	Peak wet-weather flow
Pycnocline	A region of rapid vertical change in the density field of the seawater column
QA/QC	Quality Assurance and Quality Control
RAS	Return Activated Sludge is continuously returned to the aeration tank to provide a seed of developed bacteria for rapid digestion of incoming wastewater. Some sludge is wasted (WAS) to prevent excessive solids buildup.
RFP	Request for Proposal
ROV	Remotely Operated Vehicle
RPA	The Reasonable Potential Analysis procedures in Appendix IV of the California Ocean Plan provide a quantitative methodology for statistically analyzing historical effluent data to determine the likelihood of exceeding permit limits in the future, and thus, whether continued monitoring is warranted.
RWQCB	State of California Regional Water Quality Control Board - Central Coast Region
SAIC	Science Applications International Corporation
SBE	Sea-Bird Electronics, Inc.
SCBFMC	Southern California Bight Field Methods Committee
SCBPPFCT	Southern California Bight Pilot Project Field Coordination Team
SCCWRP	Southern California Coastal Water Research Project
SEP	Supplemental Environmental Project

LIST OF ACRONYMS AND DEFINITIONS

(Continued)

Sigma-T	σ_t units are a measure of the density of water equal to the 1000 times the specific gravity minus one.
SJCF	San Joaquin Composting Facility located in Kern County California and owned by McCarthy Family Farms, Inc.
SLO EHS	<u>San Luis Obispo County Environmental Health Services</u>
SOP	Standard Operating Procedure
STLC	Soluble Threshold Limit Concentration applies to the measured concentration in the liquid extract from a biosolid sample, as determined by a Waste Extraction Test. The State of California classified biosolids with leachate concentrations exceeding the STLC as hazardous.
STORET	STORage and RETrieval electronic data system for water quality monitoring data developed and administered by the USEPA
SWRCB	State Water Resources Control Board of the California Environmental Protection Agency
Thermocline	A region of rapid vertical change in the temperature field of the seawater column
TIC	Tentatively Identified Compound is a compound that can be seen by the analytical testing method, but its identity and concentration cannot be confirmed without further analytical investigation. Many analytical methods can report TICs but the analysis is not targeted specifically for their detection.
TKN	Total Kjeldahl Nitrogen
TRC	Total Residual Chlorine in effluent is determined from grab samples collected downstream of the chlorine contact chamber.
TSS	Total Suspended Solids
TTLC	Total Threshold Limit Concentration applies to the total wet-weight concentration of a contaminant within a bulk biosolid sample consisting of the entire millable solid matrix rather than just the leachate. Biosolids are designated as hazardous wastes in the State of California if measured bulk concentrations exceed the TTLC.
TUa	Acute Toxicity Units
TUc	Chronic Toxic Unit
TVS	Total Volatile Solids
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGPO	United States Government Printing Office
USMMS	United States Minerals Management Service

LIST OF ACRONYMS AND DEFINITIONS

(Continued)

VFD	Variable-Frequency Drives are electronic-motor-speed controllers used at the MBCSD WWTP to precisely control the influent pumping rate at the plant headworks by adjusting the frequency of electrical power sent to the pump motors.
WAS	Waste Activated Sludge is removed from the treatment process to prevent excessive solids buildup.
WERF	The Water Environment Research Foundation is a nonprofit organization that helps utilities and corporations manage water resources.
WET	Waste Extraction Tests measure the soluble leachate or the extractable amount of a substance contained within a bulk sample of biosolids. A WET is indicated if the bulk wet-weight concentration of a contaminant in a biosolids sample exceeds ten times the STLC.
WQX	Water Quality Exchange framework providing the main mechanism for submitting data to the USEPA STORET Data Warehouse
WRRF	Water Resource Recovery Facility
WRF	City of Morro Bay Water Reclamation Facility
WRFCAC	City of Morro Bay Water Reclamation Facility Citizen Advisory Committee
WWTP	City of Morro Bay-Cayucos Sanitary District Waste Water Treatment Plant
ZID	The Zone of Initial Dilution is a limited volume of water surrounding the outfall where wastewater rapidly mixes with receiving waters. Most receiving-water objectives of the Ocean Plan do not apply within the ZID.

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EXECUTIVE SUMMARY

The City of Morro Bay and the Cayucos Sanitary District (MBCSD) jointly own the wastewater treatment plant operated by the City of Morro Bay. The treatment plant discharges effluent to the open ocean environment of northern Estero Bay under the authority of National Pollutant Discharge Elimination System (NPDES) permit No. CA0047881. The current permit was issued in January 2009. The permit allows discharge of blended primary- and secondary-treated wastewater, although the vast majority of the wastewater receives secondary treatment. The discharge permit requires a monitoring and reporting program that evaluates short- and long-term effects of the effluent discharge on receiving waters, benthic sediments, and infaunal communities. This 2016 Annual Report partially satisfies those reporting requirements.

This document presents a comprehensive analysis of the extensive monitoring data collected over the last three decades. Virtually every aspect of the treatment process, receiving waters, and seafloor sediments is monitored. An exhaustive quantitative analysis of all measured parameters demonstrates that the effluent discharge consistently meets the permit requirements and has no discernible effect on the ocean environment. A comparison of the properties of the influent and effluent affirms the treatment plant's proficiency at removing contaminants and reducing organic loads within the wastewater stream. All offshore water-quality measurements indicate that the effluent plume was largely restricted to a narrow 15-m zone of initial dilution (ZID) around the outfall. Measurements within the effluent plume collected a few meters from a diffuser port quantified the plume's rapid dispersion, and demonstrated that the structure was operating better than predicted by modeling. Finally, the absence of adverse discharge-related impacts to the physics, chemistry, and biology of benthic sediments verified the effectiveness of the treatment process, the high dilution of effluent within receiving waters, and the low toxicity of the discharged effluent. The additional data presented in this report are consistent with conclusions based on historical data insofar as the treatment plant's continued low emission of contaminants, low toxicity of the effluent stream, and absence of impacts to the marine environment.

The plant operates under a 301(h)-permit that modifies the general NPDES requirements on suspended-solid and biochemical oxygen demand (BOD) emissions alone. All other NPDES limits, including restrictions on the discharge of toxic substances, apply to the MBCSD discharge without exception. Regardless, the partial-secondary treatment currently performed by the plant routinely achieves reductions in suspended solids and BOD that are close to, and often exceed, secondary treatment requirements.

Effluent monitoring throughout 2016 documented another year of high operational performance by the treatment plant. Major effluent constituents, including suspended solids (TSS), BOD, and oil and grease (O&G), all had much lower concentrations and mass emissions than the permitted maximums, as has consistently been the case throughout the entire operational history of the plant. Although the treatment process efficiently removed major organic wastewater constituents such as suspended solids, BOD, O&G, and coliform bacteria, the general lack of other chemical contaminants within effluent was largely due to their absence within the influent stream. Like most publicly owned treatment works, the MBCSD plant is not designed to extract heavy metals and synthetic organic compounds from wastewater. Instead, the MBCSD effluent's low toxicity is primarily due to the lack of heavy industry within its service area. The few businesses that discharge to the sewer system produce wastewater that is similar to that of domestic sources, only on a larger scale. A digital database, maintained by collections system personnel, documents these sources. In addition, an ongoing public-outreach program and the convenience of an onsite household hazardous waste recycling facility further reduced the introduction of pollutants into the waste stream.

Throughout more than three decades of operation, the treatment plant has consistently outperformed expectations for wastewater treatment based on regulatory standards. During that time, there has been no indication of deterioration in plant performance, and effluent quality has consistently exceeded the performance criteria anticipated in the original design. On rare occasions where exceptions to standards or criteria have occurred, they have been the direct result of unforeseen external events, or the brief, unavoidable mechanical failure of a treatment-system component. In light of their diligent adherence to a program of preventative maintenance, the main challenge for plant personnel has been to respond quickly to unanticipated failures in system components or unforeseeable events.

Among the thousands of samples and measurements collected as part of the monitoring program during 2016, there was only one exception to the waste-discharge requirements specified in the NPDES permit. This near-perfect level of compliance is laudable and represents the culmination of many years of hard work by dedicated and experienced MBCSD personnel. Their knowledge and experience regularly enables them to enact appropriate proactive measures that ensure the smooth operation of this facility.

Quarterly offshore water-column surveys evaluated receiving-water quality. The high-precision measurements collected during the surveys were capable of resolving minute changes in ambient water properties. These measurements detected and delineated effluent as it mixed with receiving waters upon discharge from the diffuser. Slight anomalies in water properties associated with the presence of dilute effluent were observed very close to the diffuser structure within the ZID. These highly localized anomalies were still within the limits specified in the NPDES permit, even those collected well within the ZID where such limits do not apply.

The physics, chemistry, and biology of benthic sediments around the outfall have also been monitored for 31 years. As has been the case throughout the more than three decades of monitoring, all measurements fully complied with requirements of the discharge permit and the objectives of the California Ocean Plan. Benthic environments are important indicators of the presence of marine pollution because they act as a major reservoir for most contaminants that enter the ocean. The lack of perceptible impacts to the benthic environment around the diffuser structure during 2016 confirmed that the treatment process effectively removed contaminants from the influent stream and that the diffuser structure efficiently diluted wastewater upon its discharge into receiving waters.

Three sediment-chemistry analyses document the absence of discharge-related benthic impacts. First, chemical concentrations measured within Estero Bay sediments during 2016, and in prior years of monitoring, were below thresholds identified as toxic to marine organisms. This includes samples collected close to the diffuser structure, as well as those collected more than 1 km away. In fact, measured concentrations were comparable to those found in benthic sediments collected throughout the region, but they were generally much less than those found within the Southern California Bight. Second, benthic samples collected during 2016 did not exhibit significant gradients of increasing contaminant concentration with increasing proximity to the outfall. Finally, there is no evidence of a buildup of wastewater constituents near the outfall over time, based on a comparison of the long record of measurements collected near and far from the discharge.

Infauna residing within seafloor sediments also serve as indicators of marine contamination because of their limited mobility and well-defined responses to pollution. Numbers of species, abundance, biomass and other parameters describing infaunal communities can indicate the presence of contaminant-induced stresses if, for example, gradients extending from a pollutant source to distant, unaffected areas is observed. More than 262,000 infaunal organisms have been collected and examined since the beginning of the benthic monitoring program 31 years ago. Throughout the monitoring program, there has never been an indication of discharge-related impacts to benthic biota. Instead, the data have revealed a

consistently healthy indigenous infaunal community, with uniformly high diversity that does not decline with proximity to the diffuser.

In the 29 years prior to 2015, analyses demonstrated that the sediments surrounding the outfall supported an extraordinarily healthy marine community dominated by suspension-feeding organisms living in clean sediments. These observations held true despite widespread and significant temporal variation in the abundance of individual organisms. However, a major change in the benthic environment occurred in 2015 and 2016 when numerous large sand dollars dominated the benthic environment throughout the survey area. They displaced most of the resident suspension-feeding infauna and introduced opportunistic detritus feeders. Most measures of the health of the infaunal community declined as a result, including density, diversity, species counts, and richness. The concomitant decline in feeding indices were reminiscent of organic enrichment that had been observed within seafloor sediments immediately surrounding large ocean discharges in other regions prior to improvements in wastewater treatment during the 1980s.

However, the declines in the measures of infaunal wellbeing that were observed during the 2015 and 2016 surveys were unrelated to benthic degradation caused by the discharge of organic contaminants from the MBCSD outfall. First, organic loading within wastewater discharged from the MBCSD outfall during 2015 and 2016 was among the lowest on record. Second, the declines occurred uniformly throughout the offshore survey area with no evidence of a spatial gradient related to outfall proximity. Third, the concentrations of organic constituents measured within sediment samples collected at the infaunal sampling sites during 2015 and 2016 were comparable to those of prior years. Instead, the observed decline was directly associated with the establishment of a mature sand dollar bed. During 2016, the aging sand dollar population began to die off, and introduced isolated pockets of decaying organic matter that were quickly populated by opportunistic detritus feeding organisms. These organisms introduced level of infaunal variability during the 2016 survey that was unparalleled in the three-decade-long benthic monitoring record.

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1.0 INTRODUCTION

The City of Morro Bay and the Cayucos Sanitary District (MBCSD) jointly own the wastewater treatment plant operated by the City of Morro Bay. A 301(h)-modified National Pollutant Discharge Elimination System (NPDES) permit was originally issued to the MBCSD in March 1985 (Permit Number CA0047881). Region IX of the U.S. Environmental Protection Agency (USEPA) and the Central Coast California Regional Water Quality Control Board (RWQCB) jointly issued the permit. Following extensive evaluation processes, the USEPA and the RWQCB reissued the permit three times: March 1993 (RWQCB-USEPA 1993ab), December 1998 (RWQCB-USEPA 1998ab), and January 2009 (RWQCB-USEPA 2009ab). This report addresses compliance with the provisions of the most recent permit.

Marine Research Specialists (MRS) began conducting the Offshore Monitoring and Reporting Program for the MBCSD in July 1993. Since then, the treatment plant staff, MRS, and various subcontracted laboratories, have collected and analyzed samples as part of the monitoring program.

1.1 REPORT ORGANIZATION

This 2016 annual report summarizes results from the four major monitoring components that analyze the treatment plant, receiving waters, marine sediments, and benthic biota. The organization of this document is similar to that of previous annual reports for continuity with these four reporting aspects of the monitoring program. Other than this brief introductory chapter, the major sections include the following:

Chapter 2	Treatment Plant Performance
Chapter 3	Receiving Water Quality
Chapter 4	Bottom Sediments and Biota
Chapter 5	Conclusions and Recommendations

Chapter 2 details all aspects of the onshore portion of the monitoring program, including the operation and performance of the treatment plant and the disposition of biosolids during 2016. Chapter 3 documents the four offshore surveys that were conducted during 2016 to assess the quality of ocean waters where the effluent is discharged. Chapter 4 documents the analysis of sediment samples collected during an offshore survey in October 2016. It places the chemical and biological measurements in historical context by incorporating time-series analyses of the entire 31-year database. Finally, Chapter 5 reiterates the major conclusions of this report, and presents recommendations for future actions concerning the monitoring program and operation of the treatment plant.

In addition to these chapters, a large set of appendices provides detailed supporting documentation for the analyses in the body of this report. Appendix A lists the design specifications of the wastewater treatment plant. Appendix B details the biological analysis of the offshore sediment samples and contains information about the taxonomists who performed the species-level identification of the specimens. Appendix C contains benthic sample descriptions, including the location, depth, and grain-size characteristics of sediment samples. Appendix D contains a report of the annual outfall inspection that was conducted by diver. The original benthic chemistry report and quality assessment and quality control (QA/QC) analyses performed are provided in Appendix E. QA/QC procedures for the analytical laboratories are provided in Appendix F. This appendix also contains a listing of updates to the MBCSD treatment plant operations manual that were completed during 2016, and a summary of the analytical methods used by BC Laboratories in their chemical analyses of the samples.

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2.0 TREATMENT PLANT PERFORMANCE

During 2016, the treatment plant removed nearly all organics and other solids from the wastewater stream. Analyses of key diagnostic constituents, including suspended solids, biochemical oxygen demand (BOD), and oil and grease (O&G), documented the high operational performance of the plant. Similarly, periodic analyses of effluent for trace metals, pesticides, priority pollutants, and toxicity demonstrated the benign environmental character of the effluent. A proactive operation and maintenance (O&M) program minimized the number of avoidable exceptions to the limits specified in the NPDES discharge permit during 2016. The sections below use these analyses to evaluate regulatory compliance through a comparison between the measured wastewater characteristics and the limits cited in the NPDES discharge permit issued to the City of Morro Bay and Cayucos Sanitary District in January 2009 (RWQCB-USEPA 2009).

Data collected during 2016 augment three decades of prior monitoring information that chronicle a treatment process that consistently achieves a high level of performance. As in previous years, removal of suspended solids and oxygen-demanding materials during 2016 significantly surpassed permit-required minimum limits. In many instances, the treatment plant's performance approached or exceeded the standards for full secondary treatment.

Despite processing an unusually large fraction of sewage in the wastestream during 2016, the plant discharged only 21% of the allowed solids, while still attaining removal rates 15% higher than the permitted minimum. The treatment plant removed solids at a rate exceeding the 85% standard for full secondary treatment in all months except January, when the removal rate was immaterial because influent solids concentrations were already low. Additionally, the average effluent suspended-solids concentrations during four months of the year remained at or below the 30-mg/L criterion for full secondary treatment. The annual average effluent suspended-solids concentration (37 mg/L) was only about half of the permitted maximum (70 mg/L), and the total solids emission (42 MT¹) was only about one-fifth of the allowed solids discharge (199 MT).

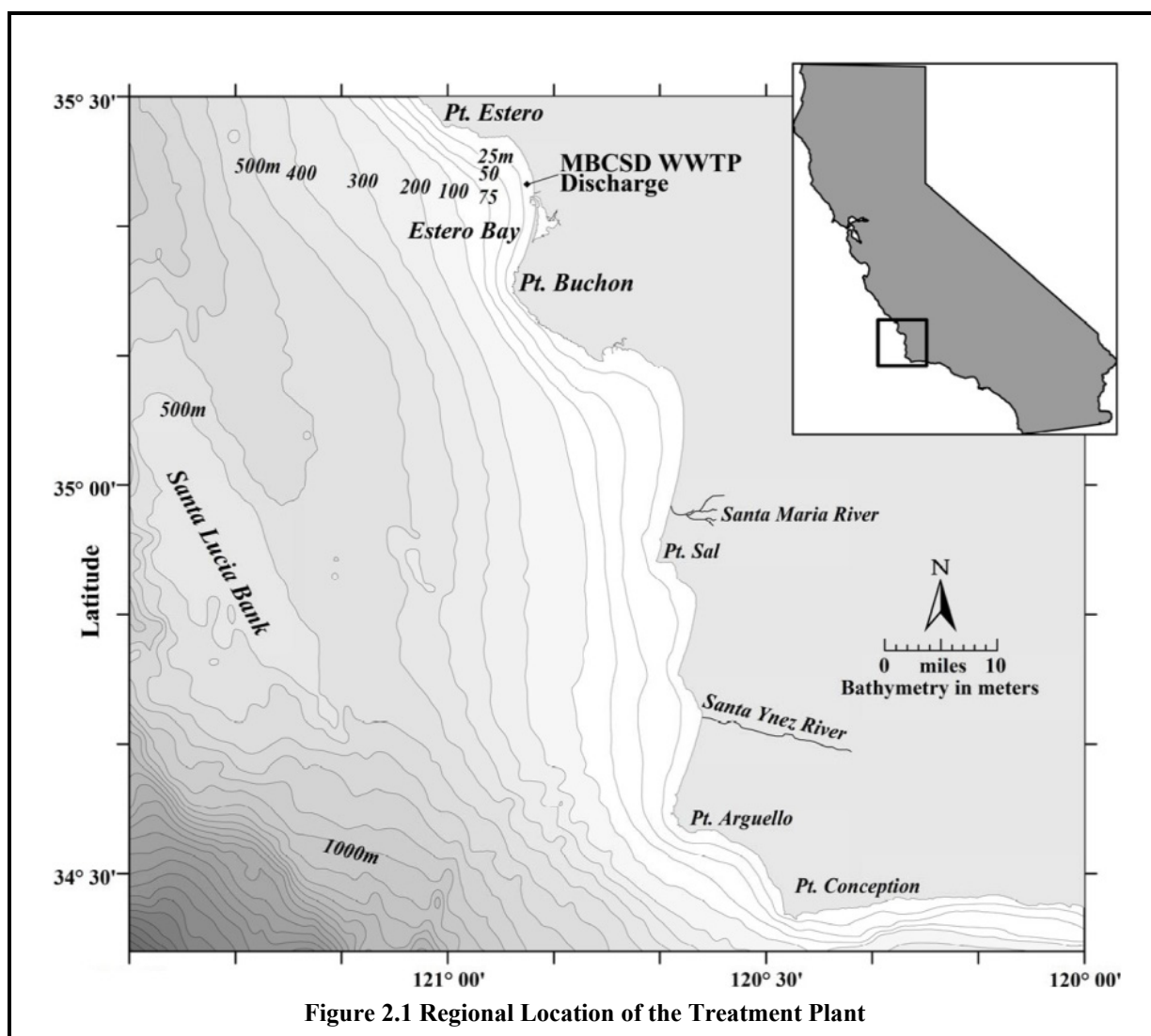
During 2016, the treatment plant also removed the vast majority (83%) of oxygen-demanding material from the influent stream. Technology-based requirements for BOD are generally unimportant for open-ocean discharges because they are unlikely to result in oxygen depletion (Page 6 in National Academy of Sciences 1993). Nevertheless, the average BOD removal rate met or exceeded the 85% monthly standard for full secondary treatment during three months of the year, while the lowest monthly removal rate of 72% was nearly 2½ times greater than the minimum permitted rate of 30%. Additionally, the annual average effluent BOD concentration (64 mg/L) was only about half of the permitted maximum (120 mg/L), and the total mass emission (74 MT) was about one-fifth of the allowed BOD discharge (342 MT).

The general absence of industrial contaminants in the wastestream attests to the benign nature of the influent, which is almost entirely generated by nonindustrial residential sources. Chemical analyses for 78 chemical compounds quantified low-level concentrations of only of only two common wastewater constituents and three naturally occurring trace metals. All measured concentrations were well below permit limitations. Chronic bioassays conducted on effluent samples that were also collected twice during 2016 determined that the discharge had very low toxicity to marine organisms, a result consistent with the prior 22-year record of toxicity testing.

¹ Metric tons or 1,000 Kg (1.1 short tons)

2.1 WASTEWATER TREATMENT PLANT

The Morro Bay/Cayucos Wastewater Treatment Plant (WWTP) is publicly owned and operated by the City of Morro Bay and the Cayucos Sanitary District (MBCSD). The WWTP is located in the City of Morro Bay, within San Luis Obispo County, along the central coast of California (Figure 2.1). The plant serves the Morro Bay and Cayucos communities, which, according to the 2010 census, have a combined population of approximately 12,835 (10,243 in Morro Bay and 2,592 in Cayucos). The WWTP discharged, on average, 0.84 million gallons per day (MGD) during 2016, which is by far the lowest on record for the three-decade-long monitoring history. The plant was designed to accommodate an average dry-weather flow of 2.06 MGD, a peak seasonal dry-weather flow (PSDWF) of 2.36 MGD, and a peak wet-weather flow (PWWF) of 6.64 MGD. The plant's flow did not even approach, much less surpass, any of these design limits during 2016.



2.1.1 Operations

Throughout most of 2016, eight trained personnel operated the WWTP (Table 2.1). Bruce Keogh served as both the Wastewater Division Manager and the Laboratory Director; and Les Girvin continued to serve as WWTP Supervisor. However, at the end of the year, a number of individuals retired, and John Gunderlock was promoted to Chief Plant Operator. Additionally, a number of new personnel were recruited in early January 2017 to replace staff that had recently retired.

WWTP personnel provided laboratory workspace to the National Estuary Program (NEP), which is dedicated to protecting and restoring the natural resources of Morro Bay and its watershed. NEP volunteers used the WWTP laboratory to analyze bacterial samples collected throughout the Morro Bay watershed.

**Table 2.1 Morro Bay/Cayucos WWTP
Personnel During 2016**

Name	Term	Grade and Certification No.
Bruce Keogh	1/1–12/22	IV – 8978
John Gunderlock	1/1–12/31	V – 10500
Les Girvin	1/1–12/30	III – 8499
Richard Fernandez	12/12–12/31	III – 41341
Steven Aschenbrener	1/1–12/31	II – 7548
George Helms	1/1–12/22	II – 28158
David Bierman	1/1–12/22	II – 28745
Neza Chavira	1/1–8/5	II – 40669
Dane Lundy	1/1–12/31	II – 36547
Adam Hegg	1/1–4/7 11/16–12/31	I - OIT

WWTP personnel attended workshops, seminars, and continuing education classes throughout 2016. Training courses and seminars covered topics such as Environmental Safety Training, Confined Space Entry, Trench Shoring Safety, Lockout/Tagout Electrical Hazard Control, and OSHA¹ Construction Safety Training. Laboratory personnel training included refresher coursework in Basic Water Analysis, Laboratory Exam Review, and Distribution System Field Monitoring and Chlorine Testing for Small Systems. Other education-related efforts included visits to other treatment plants in the region, including the Los Osos WRF.² Separate tours of the Pismo Beach and Paso Robles Wastewater Plants were dedicated to an exchange of information on data management, including protocols for mandatory monitoring-data submissions to CIWQS.³ Additional discussion centered on plant and collection-system management and staffing levels.

WWTP personnel also conducted plant tours for the public, students from local schools, members of various local agencies, staff from other treatment plants, and candidates for future employment at the WWTP. Concern citizens and members of the WRFCAC⁴ learned about treatment-plant processes and gathered information about the City's plans for a new WRF during a number of tours of the WWTP that were conducted throughout the year. Staff from the City of Modesto Wastewater Division focused on the design and operation of the WWTP influent screens during their tour. Regulatory-based tours were conducted on separate occasions for RWQCB and APCD⁵ staff in conjunction with their permit-inspection responsibilities.

Plant operations during 2016 included ongoing maintenance of an overall health and safety plan. The WWTP passed an annual inspection by SLO EHS⁶ in conjunction with the WWTP Hazardous Waste Business Plan on file with that county agency. The WWTP's chemical response plan and confined-space entry protocol was reviewed with a representative of the California Joint Powers Insurance Authority during his tour of the WWTP.

¹ Occupational Safety and Health Administration

² Water Reclamation Facility

³ California Integrated Water Quality System Project

⁴ City of Morro Bay Water Reclamation Facility Citizen Advisory Committee

⁵ San Luis Obispo County Air Pollution Control District

⁶ San Luis Obispo County Environmental Health Services

In addition to performing the periodic effluent analyses required by the NPDES Permit during 2016, the laboratories involved in the analyses of the MBCSD WWTP samples participated in laboratory performance evaluations intended to evaluate the accuracy of effluent measurements and ensure the overall quality of the monitoring reports. In particular, the adequacy of each laboratory's analytical chemistry capabilities was demonstrated during 2016 when acceptable results were achieved in a Water Pollution Proficiency Testing Study, which is an annual requirement of laboratories certified by the State of California Environmental Lab Accreditation Program (ELAP) and is sponsored by the State Water Resources Control Board and the USEPA. Successful completion of the federally mandated Discharge Monitoring Report Quality Assurance Study 36 (DMR-QA 36) satisfied this state regulatory requirement.

The MBCSD WWTP laboratory analyzed wastewater parameters reported on a daily and weekly basis, including residual chlorine, suspended solids, turbidity, coliform, pH, and BOD. Aquatic Testing Laboratories evaluated the toxicity of effluent to marine organisms with semiannual bioassays. BC Laboratories, Weck Laboratories, Vista Analytical Services, and Monterey Bay Analytical Services determined the concentrations of a wide range of chemical compounds within effluent on a semiannual basis. BC Laboratories also performed the annual chemical analysis of biosolid and benthic-sediment samples, as described in Section 2.3 of this chapter, and in Chapter 4. Each laboratory's quality assurance and control plan is included in Appendix F of this report.

In recent years, both the City of Morro Bay and the Cayucos Sanitary District actively sought to identify and eliminate sources of inflow (rainwater runoff) and infiltration (of groundwater), known as I&I, within their respective collection systems. Reducing the volume of I&I entering the collection system minimizes the amount of non-wastewater unnecessarily processed by the WWTP. For example, collection system staff cleaned over 175,000 feet of sewer pipe, video inspected mainline sections, and completed four spot repairs to the collection system at various locations throughout Morro Bay. Mainline repairs are instituted to eliminate root infestations and other sources of possible infiltration. In prior years, many conventional manholes throughout Morro Bay were replaced with sealing manhole lid/ring combinations, and some manholes have been epoxy-sealed to reduce I&I further. Finally, collections personnel occasionally conduct smoke testing to identify potential sources of I&I further, although smoke testing was not performed during 2016.

Meanwhile, at the WWTP itself, personnel maintained a proactive preventative maintenance program that included major preemptive repairs of various components of the treatment process and the outfall. In recent years, inspection and repair of critical plant components was planned, budgeted, and completed as part of a Major Maintenance and Repair Plan (MMRP). The scope of MMRP projects exceeded normal maintenance activities, encompassed entire treatment components, and occasionally required major infrastructure reconstruction. Based on the success of the MMRP, which addressed the repair of most major components of the treatment process and provided a positive overall assessment of current plant condition, the MMRP was phased-out during 2016. During the latter half of 2016, more routine inspection, cleaning, repair, and replacement of individual pieces of treatment equipment within the WWTP has been identified and budgeted as part of the ongoing O&M¹ effort. In future years, only those O&M projects necessary to keep the WWTP operating efficiently, safely, and within regulatory compliance during transition to the new wastewater facilities will be emphasized. Currently, fifteen O&M projects have been identified for attention in 2017. In the event that the operational timeframe of the WWTP requires extension and major treatment components begin to reach the end of their service life, the MMRP could be reinstated.

¹ Operations and Maintenance

To assist WWTP staff in the O&M of major plant components, an O&M manual (dated November 1987) was completed following the plant upgrade in 1986. The O&M manual includes standard operating procedures (SOPs) as well as preventative and contingency plans for controlling and minimizing the effects of accidental discharges. The manual is reviewed and updated as new major equipment is placed online or when procedures are changed. A list of revisions and updates to the manual that were completed in 2016 are included in Appendix F.2 of this report.

As part of routine maintenance procedures, divers inspect the ocean outfall's exterior and the diffuser structure for signs of damage annually. The outfall was inspected on September 13, and the written inspection report is included herein as Appendix D. The outfall and diffuser system was found to be in good condition, with no broken or plugged diffuser ports. The divers provided photodocumentation of the outfall's condition in the form of digital video recorded throughout their inspection. On September 14, the divers revisited the outfall site to replace the two spar buoys that mark the ends of the diffuser structure. The associated ground tackle used to anchor the buoys was also replaced.

Figure 2.2 is a photograph of a diffuser port taken during a previous outfall inspection. It shows a dense cover of marine epifaunal organisms thriving on the outer surface of a diffuser port. A large colony of club-tipped anemones (*Corynactis californica*), bright pinkish-red in color, covers the top surface of the port. The continued presence of these filter-feeding organisms attests to the benign nature of the effluent discharge, and to the outfall's value as an artificial reef. Quantitative biological surveys conducted within the region found that these anemones are only occasionally observed on high-relief rock surfaces within Estero Bay, and then only in deeper water (>85 m) (Morro Group 1999). Ostensibly, their susceptibility to elevated suspended-sediment loads explains their rarity on nearshore, lower-relief rocky substrates.

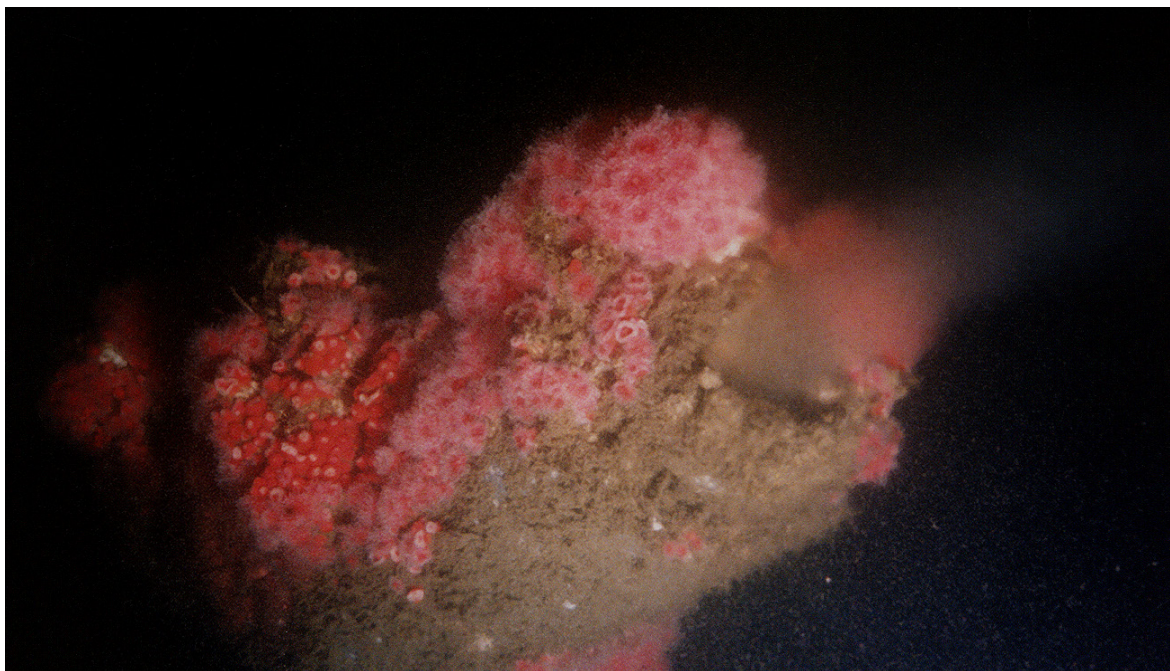


Figure 2.2 Photograph of a Diffuser Port Discharging Effluent

In addition to the routine calibration, maintenance, and repair efforts conducted at the WWTP, a number of significant repairs were completed or initiated in 2016, both as part of the MMRP phase-out and the ongoing the O&M program. Monthly operations summaries documented these efforts throughout 2016 (MBCSD 2016a thru 2016l). These narrative reports were attached to the monthly monitoring-data submissions to CIWQS.

As part of the MMRP wrap-up in 2016, the last of the three sludge digesters at the WWTP was repaired and rehabilitated. Digesters #1 and #2 had already been refurbished in prior years, and Digester #3 was taken offline for several months during 2016 to perform major reconditioning. In addition to other rehabilitation activities, its corroded gas line was replaced, which required careful preparation to limit atmospheric release of digester gas. This included operational changes to biosolids process, and preparation of the gas line and manifold by nitrogen purging and dry-ice packing. The APCD was notified of the activity prior to, and after the completion of the work. Digester maintenance was completed in May, and Digester #3 was brought on-line in a controlled manner to monitor and ensure its performance. Initially, it operated as the secondary digester, but at the end of June, it began processing primary sludge, initially at a limited feed rate.

One of the more important O&M activities conducted in 2016 focused on the Chlorine Contact Tank. Although this critical component of the complex disinfection process received significant improvements in 2015 as part of the MMRP, organic solids slowly accumulate on the Tank's floor, and the equipment within the Tank is sensitive to damage or failure that is hidden from view. Because of this, the Tank is now regularly drained, inspected, washed, and repaired even though the draining procedure requires significant preparation and effort to avoid discharge of effluent with unacceptable levels of either chlorine or bacteria.

During 2016, the Chlorine Contact Tank, and/or its adjacent vaults and chambers were drained on seven separate occasions. Draining is an involved process that requires a temporary interruption in effluent discharge. To limit impacts on the overall treatment process from flow interruption, Tank draining and repair is conducted very early in the morning, when plant flow is lowest. In preparation for stopping flow to the Tank, the Plant's Grit Chamber and a Primary Clarifier are drained so that they can act as temporary detention basins. The contents of the Chlorine Contact Tank are then emptied into sludge beds for temporary storage and later treatment. Lastly, as inflow begins to fill the temporary detention basins during the repair, the influent pumps are shut down and influent is allowed to backup in the trunk line for later treatment once the Tank is back online.

The chlorine dose-response system was another aspect of the disinfection process where careful monitoring was required throughout 2016. WWTP personnel responded to chlorine-residual dial-out alarms on 30 separate occasions. These alarms trigger on a 24/7 basis when residual-chlorine concentrations drift beyond the high or low set points on the auto analyzers. Despite frequent false alarms, a narrow range in the alarm set-points is retained because it affords enough lead-time for operators to intervene directly in the chlorination process, if needed to prevent unacceptable excursions in residual chlorine or bacterial levels. None of the 30 alarms during 2016 required a major modification to the disinfection process, but the entire disinfection dose-response system is complex (see Sections 2.2.8 and 2.2.9) and has required prompt direct intervention by operators in the past.

Other noteworthy O&M activities conducted in 2016 are itemized below.

- Installed new flow meters on sludge pumping lines to improve the accuracy of instantaneous and totalized flow readings;
- Constructed masonry walls around the periphery of the two biofilters to mitigate potential inundation from rainwater flow into the WWTP; and
- Installed a new level-control device in the influent channel at the headworks to provide a more accurate water-surface elevation signal to control the main influent pumps when the elevation approaches maximum level.

2.1.2 Plant History

The original WWTP, built in 1954, had a nominal capacity of 0.7 MGD and a 1-MGD maximum throughput. The original plant included a headworks structure, primary and secondary clarifiers, a biofilter, a single-stage digester, chlorination facilities, biosolids drying beds, and a short ocean outfall.

In 1964, the plant upgraded to a nominal capacity of 1 MGD and a 1.3-MGD maximum throughput to meet the demands of the growing coastal community. This upgrade added a pump station, a splitter box, a primary clarifier, a secondary clarifier, a biofilter, chlorination facilities, biosolids beds, and another primary digester, which allowed conversion of the existing digester to process secondary sludge. The existing office building and laboratory were also constructed during this upgrade.

During the 1970s, the City of Morro Bay developed a plan for additional upgrades to the WWTP facilities intended to augment the plant's capacity further. In 1980, the City began designing these planned improvements, including the construction of a new outfall to protect the marine environment, and an upgrade of processing equipment to provide full secondary treatment.

Following a yearlong study of oceanographic conditions within Estero Bay, design of the new outfall was completed in April 1981. The new outfall and diffuser system extended the discharge from the surfzone to a point much farther offshore. This deeper discharge significantly increased the effluent mixing rate within the open-ocean environment. The new outfall was completed and placed in service in June 1982.

The design of the facility improvements, completed in September 1981, called for a final effluent suspended-solids concentration of 30 mg/L and an equivalent limit on BOD. However, monetary aid from state or federal agencies to finance the construction of a full-secondary-treatment facility was not available. Since discharge through the new outfall was not causing any apparent adverse environmental impacts, and the projected future throughput was low, the State determined that additional financial aid for upgrading the MBCSD WWTP to full secondary was not warranted. Instead, the City modified the design to provide secondary treatment to a majority (1 MGD) of the projected flow to ensure full compliance with the state water-quality standards set forth in the California Ocean Plan (COP) (SWRCB¹ 1990).

State officials concurred with this partial-secondary level of treatment, provided that the USEPA approve a 301(h)-modified NPDES discharge permit that adjusted secondary-treatment requirements on suspended-solids and BOD emissions. In a March 1983 letter, RWQCB staff formally determined that the proposed discharge would comply with state water-quality standards pursuant to Subsection 301(h) of Title 40 of the Code of Federal Regulations.²

¹ State Water Resources Control Board of the California Environmental Protection Agency

² Section 125.60(b)(2) [40 CFR 125.60(b)(2)] (USGPO 1982a) and 40 CFR 125.63(b) of the 301(h) regulations dated November 1982 (USGPO 1982b)

Upgrades to the treatment plant, completed between 1983 and 1985, increased the plant's capacity to a 2.06-MGD average dry-weather flow and a peak flow of 6.6 MGD. The plant now includes primary treatment of all influent by screening, grit removal, and primary sedimentation. Additionally, depending on the hydraulic conditions within the plant, up to 1 MGD of the flow can be diverted through a secondary-treatment process consisting of trickling filters, clarifiers, and a solids-contact chamber. The secondary-treatment process utilizes two trickling filters, an aerated solids-contact channel, and a secondary sedimentation tank. The original 1954 sedimentation tank was converted into a chlorination system where the primary- and secondary-treated effluents are mixed and disinfected prior to dechlorination and discharge through the ocean outfall.

In March 1985, an NPDES permit based on the previously approved Section-301(h) modification established water-quality standards for the MBCSD WWTP. The permit required treated effluent to achieve a suspended-solids content of no more than 70 mg/L (75% removal) and a maximum BOD of 120 mg/L (30% removal). The permit also required an extensive monitoring program to assure maintenance of environmental quality. The permit was valid for five years and expired in March 1990. After an evaluation process, the permit was reissued in December 1992. During this evaluation period, improvements to the treatment facilities included the installation of a sludge-removal system within the Chlorine Contact Tank.

The MBCSD again applied for renewal of the permit in May 1997, supporting its application with an extensive technical review of more than 10 years of monitoring data (MRS 1997a). An administrative extension until December 1998 allowed regulatory agencies additional time to review and issue the new permit (RWQCB 1998). In July 1998, RWQCB staff determined that the discharge described in the MBCSD application “*would comply with applicable state laws, including water quality standards, and would not result in additional treatment, pollution control, or other requirements on any other point or nonpoint source.*” This permit was finalized and issued by USEPA on 26 January 1999, with an effective date of 1 March 1999.

Based on discussions between RWQCB, USEPA, and MBCSD staff and their consultants, the following revisions were also implemented in the 1999 permit:

- A 12.7% reduction in the allowed mass emission of suspended solids, BOD, and O&G;
- More extensive reporting requirements for biosolids;
- Elimination of shellfish monitoring;
- A revised benthic sampling program, increasing the number of stations close to the diffuser structure and eliminating seasonal sampling;
- A revised receiving-water sampling program, doubling the number of vertical profiles close to the diffuser structure and eliminating bottle casts; and
- Specification of mass emission goals for toxic chemicals.

Based on the historical absence of perceptible impacts from the discharge, and the projected continuation of consistently high effluent quality, the MBCSD again applied for a renewal of the discharge permit in July 2003 (MBCSD 2003). As with the previous permits, the application requested continued discharge under the 301(h) provision that allows minor modifications to the BOD and suspended-solids requirements. In February 2004, the RWQCB (2004) administratively extended the existing permit to allow time for further review. In September 2005, the USEPA Region IX (2005) issued a tentative decision concurring with issuance of a permit to the MBCSD in accordance with Section 301(h) of the Clean Water Act. In April 2006, the USEPA and RWQCB staff issued a joint notice for a proposed action to reissue the 301(h) modified NPDES discharge permit to the MBCSD. However, in May 2006, the RWQCB and the USEPA conducted a joint public hearing addressing the reissuance of the MBCSD

permit wherein the RWQCB voted to continue the hearing pursuant to the issuance of a biological evaluation by the USEPA (2007).

In 2008, the USEPA issued an Endangered Species Act (ESA) biological evaluation of continued discharge under a 301(h) modified discharge permit wherein they determined:

...that the continued wastewater discharge from the Morro Bay/Cayucos facility is not likely to adversely affect the brown pelican or southern sea otter, both of which occur in the vicinity of the subject discharge. EPA finds that any potential direct or indirect effects of the continued wastewater discharge would be insignificant to the brown pelican and southern sea otter.

Pursuant to Section 7 of the ESA, the proposed action of the USEPA required consultation with the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS), which protect federally listed endangered species and designate critical habitat that may be affected by the proposed action. In December 2007, the USFWS concurred with the USEPA's "*determination that the proposed project is not likely to adversely affect the brown pelican or southern sea otter.*"

Nevertheless, the USEPA incorporated three conservation measures into the new NPDES discharge permit to address concerns about potential contributions to otter morbidity by cat litter and domoic acid poisoning. First, the MBCSD would implement a public outreach program to minimize the input of cat-litter-box waste into the municipal sewer system. Second, the MBCSD would be required to monitor nutrient loading from the WWTP on a regular basis. Third, the facility would upgrade to a minimum of full secondary-treatment levels by 2014.

Based on the foregoing findings and the incorporation of conservation measures, the RWQCB unanimously adopted the new discharge permit in December 2008. Subsequently, on 9 January 2009 the California Coastal Commission unanimously determined that the new discharge permit complied with the California Coastal Zone Management Act. On 14 January 2009, the USEPA issued the new NPDES permit, effective 1 March 2009. In addition to the conservation measures noted above, the following revisions were also implemented in the 2009 permit:

- Elimination of acute toxicity testing;
- Implementation of triggered shoreline coliform monitoring;
- Revision of benthic sampling pattern, eliminating cross-shore stations and shifting from grab to composite sediment-chemistry samples; and
- Revision of receiving-water sampling program, reducing the number of vertical profiles and implementing a tow-survey component.

Although the current NPDES permit was not finalized until January 2009, the MBCSD had actually begun to implement several of the proposed conservation measures years before. For example, in April 2006, the MBCSD, working to address the concerns of the USEPA and RWQCB, adopted an eight-year time schedule to rehabilitate and upgrade the treatment plant to tertiary treatment, including onsite composting, as the preferred alternative for upgrading the WWTP by 2014. The MBCSD subsequently adopted a draft facilities master plan that outlined the facilities necessary for a tertiary treatment capacity of 1.5 MGD in September 2007. Meanwhile, in August 2007 the City of Morro Bay and the Cayucos Sanitary District individually adopted revenue programs that identified increases in sewer-rate fees necessary for each community to finance the proposed plant upgrade and to provide revenue for needed sewer-system capital improvement projects. In July 2008, the City of Morro Bay implemented new residential and commercial water-use rates that increased the existing fees by 50%.

During 2009, based on the findings of flood hazard analysis, the City and District voted to relocate the treatment plant site to an elevated area adjacent to the existing treatment plant. In October 2009, the MBCSD public noticed a Request for Proposal (RFP) for Engineering Design Services for the upgrade, and the City of Morro Bay released a Revised Notice of Preparation for the project, reflecting changes to the project description involving construction of treatment-plant components adjacent to the current ones. Demolition of the existing plant was to occur after the relocated treatment-plant components were constructed and brought online. The engineering design contract was awarded at the February 2010 Joint Powers (JPA) meeting with a projected completion in 15 months.

The final Environmental Impact Report for the upgrade project was released in December 2010. The Morro Bay City Council certified the EIR¹ and approved the Conditional Use Permit and Coastal Development Permit (CDP) for the upgrade project in January 2011; however, the decision to issue the CDP was subsequently appealed to the California Coastal Commission. A *de novo* hearing was held in January 2013 at which the Commission denied the CDP for the proposed upgrade project at its current location.

Subsequently, at the February 2013 JPA meeting, the MBCSD approved development of an MMRP. The Plan was instituted to ensure uninterrupted operation of the existing WWTP in compliance with regulatory requirements during the extended operational period required for the development and construction of a new treatment facility at a different site. In August 2013, the MBCSD submitted an application to the RWQCB for a new discharge permit to replace the current 301(h) modified permit due to expire on 28 February 2014. Based on direction from the MBCSD and RWQCB staff, an application was submitted for a full secondary discharge permit.

During 2013 and 2014, the MBCSD explored various sites and treatment alternatives for a new facility to process wastewater currently treated by the existing WWTP. In early 2015, the City of Morro Bay identified a Water Reclamation Facility (WRF) as the preferred design, and initially selected a location east of the City and north of Highway 41. On April 30, the Cayucos Sanitary District suspended participation with the City in their WRF and began planning a separate Water Resource Recovery Facility (WRRF). The District began independently evaluating wastewater treatment alternatives, characterizing flow rates and mass loadings specific to the District's collection system, identifying beneficial uses for recycled water, evaluating potential facility locations, and developing a funding and financing strategy.

The WRRF is now encompassed in the Cayucos Sustainable Water Project (CSWP) that also incorporates the necessary conveyance infrastructure. During 2016, the Cayucos Sanitary District selected a membrane bioreactor as the WRRF treatment process, identified a CMAR² as the construction delivery method for the CSWP, selected the CMAR contractor, began preparing a dEIR,³ and purchased a property for the WRRF site along Toro Creek Road. The CSWP has a target completion in 2018.

At the same time, the City of Morro Bay staff and their consultants made significant progress on developing the new WRF to replace the existing WWTP. A Memorandum of Understanding was executed for the future purchase of a property near the South Bay Boulevard exit from Highway 1, after site-selection studies were performed to identify a preferred location for the new WRF. A Facility Master Plan was also completed and is presently in Draft form. The City of Morro Bay received approval for a

¹ Environmental Impact Report

² A Construction Manager at Risk (CMAR) is a project delivery method wherein the Construction Manager (CM) is required to deliver a project within a guaranteed maximum price.

³ Draft Environmental Impact Report. The CSWP dEIR was completed in February 2017 and is available at: <http://www.cayucossd.org/cswp-draft-eir>

\$10.3M loan from the SWRCB CWSRF¹ for the design and planning of recycled water projects. The WRF is scheduled for completion by May 2021.

In a February 2015 letter from RWQCB staff, they reiterated their intent to issue only one additional five-year NPDES permit for discharge from the existing WWTP. They indicated that the new permit would contain a time schedule for interim limits that would protect the MBCSD from excursions above full-secondary standards. Exceedance of secondary standards after five years from the date of permit issuance could be subject to minimum mandatory penalties. Presently, the existing NPDES discharge permit remains in force under an administrative extension pending issuance of this new permit.

2.1.3 Regulatory Setting

The 1972 Federal Clean Water Act and its 1977 amendments established national water-quality goals and created a national permit system (NPDES) of minimum standards for the quality of discharged waters (USGPO 1997a). Pursuant to the new system, states established standards specific to water bodies and designated the types of pollutants to be regulated. Since 1973 the California State Water Resources Control Board and its nine Regional Water Quality Control Boards have been delegated the responsibility of administering permitted discharges into the coastal marine waters of California. The State Board prepares and adopts the COP, which incorporates the state water-quality standards that apply to all NPDES permits. The RWQCB established a Water Quality Control Plan for the basin containing San Luis Obispo County waters ("The Basin Plan" RWQCB 1994). The Plan's standards incorporate the applicable portions of the COP, and promulgate water-quality objectives and toxic material limitations that are designed to protect the beneficial uses of receiving waters within individual basins. The Basin Plan identified the following the beneficial uses specific to the marine waters of Estero Bay, including those adjacent to the MBCSD outfall site.

- **Water Contact Recreation (REC-1)** water uses include recreational activities involving body contact with water, where ingestion of water is reasonably possible. Specific uses include, but are not limited to, swimming, wading, water skiing, skin and scuba diving, surfing, and fishing.
- **Noncontact Water Recreation (REC-2)** water uses include recreational activities involving proximity to water but not normally involving body contact with water, where ingestion of water is reasonably possible. Specific uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tide pool and marine life studies, hunting, sightseeing, and aesthetic enjoyment in conjunction with the above activities.
- **Industrial Service Supply (IND)** water uses include industrial activities that do not depend primarily on water quality, such as, mining, cooling water supply, hydraulic conveyance, gravel washing, fire protection, and oil well repressurization.
- **Navigation (NAV)** water uses include those for shipping, travel, or other transportation by private, military, or commercial vessels. The RWQCB interprets NAV as being present within any natural body of water that that has sufficient capacity to float watercraft for the purposes of commerce, trade, transportation, and pleasure.

¹ The Clean Water State Revolving Fund (CWSRF) includes a Water Recycling Funding Program administered by the SWRCB Division of Financial Assistance.

- **Marine Habitat (MAR)** water uses support marine ecosystems including, but not limited to, preservation or enhancement of marine habitats, fish, shellfish, and vegetation such as kelp, or wildlife such as marine mammals and shorebirds.
- **Shellfish Harvesting (SHELL)** water uses support habitats suitable for the collection of filter-feeding shellfish such as clams, oysters, and mussels, for human consumption, commercial, or sport purposes. Specific uses include waters that have in the past, or may in the future, contain significant shellfisheries.
- **Ocean Commercial and Sport Fishing (COMM)** water uses encompass commercial or recreational collection of fish, shellfish, or other organisms, including uses involving organisms intended for human consumption or bait purposes.
- **Preservation of Rare, Threatened, or Endangered Species (RARE)** water uses support habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened, or endangered.
- **Wildlife Habitat (WILD)** water uses support terrestrial ecosystems including, but not limited to, preservation and enhancement of terrestrial habitats, vegetation, wildlife (e.g., mammals, birds, reptiles, amphibians, invertebrates), or wildlife water and food sources.

Section 301(b) of the Clean Water Act requires publicly owned treatment works to meet effluent limitations based on secondary treatment, which is defined in terms of limits on three effluent parameters (40 CFR 133; USGPO 1997a). These limitations are:

- Total suspended solids (TSS) concentrations not exceeding 30 mg/L as a 30-day average and removal rates not less than 85%;
- BOD concentrations not exceeding 30 mg/L as a 30-day average and removal rates not less than 85%; and
- Hydrogen-ion concentration (pH) between 6.0 and 9.0.

These limits were based on the treatment capabilities of the best available technology at the time, rather than an evaluation of the treatment necessary to reduce potential environmental impacts to an acceptable level within receiving waters. Recognizing that this level of treatment may not be necessary within ocean waters, Section 301(h) was added to the Act to allow an NPDES discharge permit to modify some or all of these full secondary-treatment requirements, if certain conditions are met. The MBCSD WWTP is a combined primary and secondary-treatment facility that has operated under a Section 301(h)-modified NPDES permit (number CA0047881) since March 1985. The modifications in this NPDES permit apply only to the TSS and BOD requirements, so all other NPDES limitations apply without exception, including those for wastewater pH and toxic compounds. The modification was issued only after the MBCSD satisfied the following additional requirements:

- Demonstrate the existence of a water-quality standard specific to the pollutant for which the modification is requested (40 CFR 125.61; USGPO 1997a). The COP specifies limits on TSS and dissolved-oxygen (DO) depression, thereby establishing the relevant standards (SWRCB 2005). In January 2009, the California Coastal Commission determined that the 301(h)-modified discharge permit issued to the MBCSD complies with the State Coastal Zone Program that incorporates COP standards.
- Demonstrate that the discharge does not adversely impact public water supplies or interfere with the protection and propagation of balanced, indigenous biological populations (40

CFR 125.62). The USFWS (1998 2007) and the NMFS (1998 2004) determined that the discharge would not adversely impact threatened or endangered species, or critical habitats, pursuant to the ESA.

- Conduct a monitoring and reporting program capable of evaluating the effects of the discharge (40 CFR 125.63). The monitoring program described in this report satisfies this requirement.
- Demonstrate that the discharge will not result in any additional treatment requirements on any other point or nonpoint source (40 CFR 125.64). The highly localized footprint of the MBCSD discharge does not overlap that of other discharges.
- Determine whether the WWTP is subject to pretreatment requirements. Since there are no known sources of toxic pollutants or pesticides in the influent, the WWTP is exempt from general pretreatment requirements in lieu of a pollution prevention program. In addition, because the discharge is considered small, it is exempt from the urban pretreatment requirement (40 CFR 125.65).
- Demonstrate whether the pollution-prevention program meets the requirement for a nonindustrial source control program (40 CFR 125.66). The MBCSD pollution prevention program implements public education and source reduction programs to limit the entrance of toxic pollutants or pesticides into the treatment plant.
- Demonstrate that there will be no new, substantially increased discharges of BOD and TSS beyond those specified in the permit (40 CFR 125.67). The historically high performance of the plant process, the limited projected growth in population and industry within the service area, and the analyses provided in this report demonstrate this.
- Ensure that the WWTP exceeds the minimum requirements for primary treatment (40 CFR 125.60). The WWTP performs “*treatment by screening, sedimentation, and skimming adequate to remove at least 30 percent of the biochemical oxygen demanding material and of the suspended solids in the treatment works influent, and disinfection, where appropriate*” (40 CFR 125.58(r); USGPO 1997a).

The MBCSD WWTP is categorized as a Class III wastewater treatment facility by the Office of Operator Certification within the California State Water Resources Control Board. The Board reclassified the facility in 2001 from a Class IV facility based on the advanced treatment process and the plant’s low flow volume. A typical Class IV facility treats more than 20 MGD in the primary process, while the MBCSD plant processes a total flow of less than 2 MGD and carries out partial-secondary treatment on a large portion of the flow.

During 2015, the California Department of Public Health developed a Management Plan for Commercial Shell Fishing within Morro Bay. The Plan provides reporting guidelines in the event of a sewage spill to the Bay or adjacent ocean. WWTP staff provided comments on the Plan, and signed a Statement of Agreement concerning its implementation.

2.1.4 Description of the Treatment and Outfall System

The WWTP operating characteristics are listed in Table A.1 of Appendix A. All wastewater is treated through a primary treatment process, which includes screening, grit removal, and primary sedimentation, as shown in Figure 2.3. Typically, a portion of the flow is diverted for an additional secondary-treatment

process using biofilters, a solids-contact chamber, and a secondary clarifier. The secondary process consists of parallel single-stage, high-rate, trickling filters whose combined outflow goes to a solids-contact channel and then to a secondary sedimentation tank. When flows exceed 1 MGD, secondary-treated effluent can be subsequently blended with primary-treated effluent, before the entire blend is chlorinated for disinfection and then dechlorinated. The disinfected and dechlorinated effluent is discharged into Estero Bay through a 4,400-ft (1,341-m) outfall terminating in a multi-port diffuser system. Waste biosolids are anaerobically digested, dried, composted and used as soil conditioner and fertilizer. A schematic of the biosolids process is shown in Figure 2.14 on Page 2-36.

The location of the Morro Bay-Cayucos WWTP and outfall within Estero Bay is shown in Figure 2.4 on the following page. The treated wastewater is released into unstressed, open-ocean waters at 35°23'11"N latitude and 120°52'29"W longitude. The effluent flows through a 27-in (0.69-m) diameter outfall that extends approximately 4,400 ft (1,341 m) in a northwesterly direction. The outfall terminates in a multi-port diffuser approximately 2,700 ft (827 m) from shore. The 170-ft (51.8 m) long diffuser lies at a water depth of 50 ft (15.2 m), measured relative to the mean lower low water (MLLW) datum. Twenty-eight of the 34 available diffuser ports are currently open. The remaining six ports can be made operational if the sustained discharge exceeds 6.60 MGD.

Because of its location, the MBCSD discharge does not interfere with the maintenance of water quality and beneficial uses designated for Estero Bay (listed in Section 2.1.3 on Page 2-11). The discharge occurs in well-flushed, open coastal waters where re-entrainment or accumulation of effluent will not violate applicable water-quality standards, even if combined with pollutants from other sources. Intakes and outfalls from other publicly owned treatment works are distant from the MBCSD outfall. For example, water intake for the Morro Bay desalinization plant is from saltwater wells and not from the open ocean where the MBCSD discharge occurs. Similarly, surface discharge of water from the desalinization plant, when it does occur, is far south of the MBCSD discharge point and does not add chemical loads to the ocean environment.

2.1.5 Pollution Prevention Program

The MBCSD's Pollution Prevention Program aims to minimize the introduction of incompatible contaminants, such as pollutants and pesticides, into the treatment process. The NPDES permit requires an annual status report detailing efforts to comply with the requirements for a Pollution Prevention Program. This section serves as that report. Additionally, the current permit includes a specific requirement to document educational and outreach efforts regarding proper cat-waste disposal.

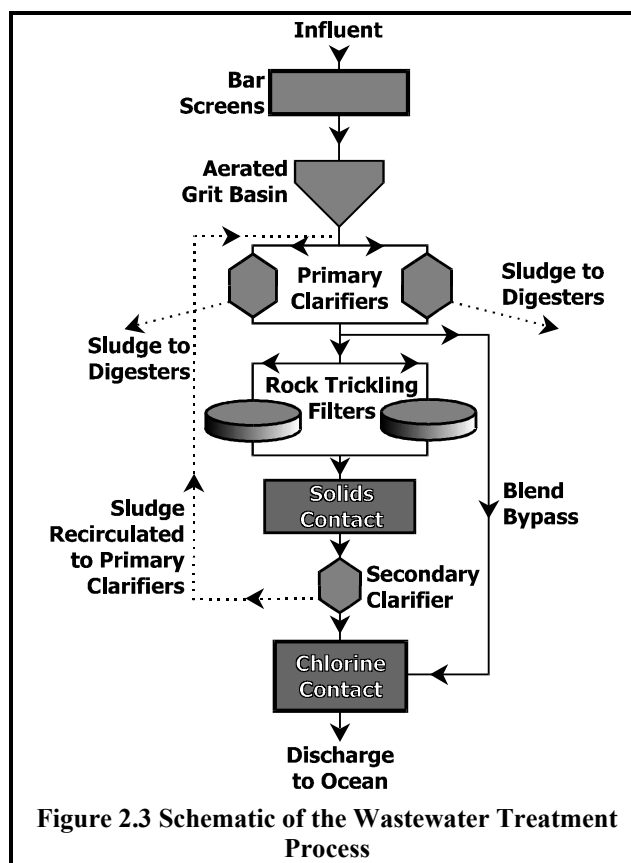


Figure 2.3 Schematic of the Wastewater Treatment Process

As in previous years, three aspects of pollution prevention were emphasized during 2016: public outreach, industrial-waste source control and identification, and monitoring of influent and effluent for industrial contaminants.

Industrial Waste Survey

During 2016, as in previous years, elevated levels of industrial pollutants were not found within the MBCSD wastewater stream. Instead, three decades of comprehensive monitoring demonstrates that effluent discharged from the MBCSD treatment plant consists of benign constituents typical of wastewater generated from domestic sources. In fact, based on analyses of past water usage, domestic sources contributed approximately 80% of the wastewater processed by the plant, with commercial businesses and government agencies contributing the remaining amount.

However, this usage-based approach overestimates the influence of nondomestic sources on the treatment process. For example, a substantial portion of the water-use attributed to the government agencies is utilized for the irrigation of landscaping, sports fields, and agricultural uses and thus, would not be expected to flow into the collection system. These agencies include the City of Morro Bay, Morro Bay High School, the San Luis Coastal School District, Morro Elementary School, and the State Department of Parks (Morro Bay State Park). Additionally, the compounds added to the wastestream by both large commercial and government users are not particularly toxic to humans or aquatic organisms and do not generally interfere with the treatment process.

This general lack of chemical contaminants within the wastestream arises because the local economy within the MBCSD service area relies largely on tourism and commercial fishing, with no heavy industry or manufacturing of any environmental significance. Nevertheless, beginning in 1999, a digital database has been used to catalogue business names, addresses, and contact information for all of the potential industrial users within the service area. This database has been used to quantify the comparatively low volume of influent derived from light industrial sources within the service area. For example, slightly more than 50 restaurants and an approximately equal number of hotels are found in the service area during any given year.



Figure 2.4 Locations of the MBCSD Outfall and Monitoring Stations within Estero Bay

The list of businesses in the database is adapted and updated regularly based on business license applications filed with the City of Morro Bay and input provided by the Cayucos Sanitary District. Businesses with no potential for industrial discharges, such as offices and retail stores, are classified separately from those with the potential for light-industrial discharge. Businesses that either do not generate wastewater at all, or discharge only domestic wastewater (e.g., theaters, beauty shops, and barbershops), are excluded from the industrial-discharge classification.

Within the City of Morro Bay, the remaining businesses are divided into two classes. Class II facilities consist of food-service establishments and constitute the vast majority of commercial businesses discharging to the collection system. Seventy-three inspections of these facilities were conducted during 2016 by the City's collection-system personnel. The inspections emphasized the best management practices (BMPs) for the disposal of fats, oils, and greases (FOG). During 2016, the City of Morro Bay updated their BMP handout and added a Spanish translation on the reverse side. During facility inspections, the new handout was distributed along with "No Grease" stickers to be displayed above sinks to raise awareness and serve as a reminder of proper FOG disposal methods. The inspections themselves focused on the presence of grease traps or interceptors, and if so, whether they were properly cleaned and maintained. For establishments with fryers, the method and frequency of oil disposal was documented.

Class 1 facilities consist of light industrial businesses such as commercial laundries, car washes, dry cleaners, print shops, and the oil-water separator maintained by the City of Morro Bay. Collections personnel perform scheduled visits, surprise onsite inspections, and formal tours of these facilities. During 2016, they inspected nine of these facilities. Car-wash discharges are considered industrial in nature because of the volume of solids, oils, and grease that are washed from vehicles. As with restaurants, the sewer-use ordinance within the City of Morro Bay municipal code requires self-service car washes to install and maintain grease traps within their sewer line connections. The municipal code also prohibits smaller contributors, like gas stations and repair garages, from disposing known contaminants into the collection system.

In addition to chemical input from light industry, the WWTP itself intentionally introduces three chemicals (ferrous chloride, sodium hypochlorite, and sodium bisulfite) into the treatment process. Ferrous chloride is used primarily to control hydrogen sulfide emissions during flaring of digester gas and heating of the digesters at the WWTP, as required by the APCD. Wastewater facilities commonly disinfect effluent prior to discharge with some form of chlorine; the WWTP uses sodium hypochlorite. However, because even low concentrations of residual chlorine can be hazardous to aquatic life, the MBCSD treatment plant adds sodium bisulfite to the wastestream to remove excess total chlorine residual once disinfection is complete.

Not surprisingly, the sodium bisulfite added to the MBCSD treatment process for dechlorination generally tracks the monthly sodium-hypochlorite dose used by the plant. Dosage also tracks seasonal changes in the fraction of organic matter entering the plant, which typically increases during the summer due to an increased tourist population within the service area, and decreased I&I. For example, during 2016, the highest hypochlorite usage (3,263 gallons) and second-highest bisulfite usage (2,874 gallons) occurred in July, reflecting a seasonal increase in the fraction of wastewater of sewage origin during the summer tourist season. Conversely, the lowest hypochlorite usage (2,484 gallons) occurred during the month of February.

Public Outreach

The MBCSD utilizes online and written literature as well as direct communication through multiple workshops, presentations, talks, and plant tours in order to educate consumers and local businesses about the organization and operation of the treatment plant; sewer-system BMPs; and techniques for the proper disposal of a variety of household wastes.

The Cayucos Sanitary District maintains a website¹ with status updates and other information relating to the CSWP, along with Town Hall Meeting notices, responses to FAQs,² and links to other resources. The City's website³ includes a series of pages devoted to an overview of the wastewater treatment plant and collection-system operations. The web pages contain pertinent information on current topics of interest. These include the status and history of the pending transition to an offsite WRF, the availability of an onsite Household Hazardous Waste Facility, and Collection System Do's & Don'ts. In addition, digital copies of all of the treatment plant's self-monitoring reports from 2005 onward, including, for example, this annual monitoring report are also available online.

The Collection System Do's & Don'ts section of the City's website also provides current BMPs for cat-litter disposal and the importance of avoiding its introduction into the collection system.⁴ This outreach program was one of the conservation measures recommended in the USEPA's (2007) biological evaluation that was conducted in preparation for issuance of the current NPDES discharge permit. During the permit-renewal process, USEPA staff postulated that minimizing the input of cat-litter-box wastes into the municipal sewer system could limit the number of *Toxoplasma gondii* parasites introduced into the marine environment, thereby mitigating a known disease vector affecting southern sea otters. RWQCB staff incorporated the USEPA's concerns into a special provision within the plant's final NPDES permit requiring the creation of a cat-litter public-outreach program.

However, immediately following the finalization of the permit, on 21 January 2009, Johnson et al. (2009) published the results of a detailed field study of southern sea otter exposure to *T. gondii*. That study unequivocally demonstrated that the incidental disposal of cat litter into the MBCSD collection system was not the cause of the observed impacts on otter morbidity from *T. gondii* infection. The authors of the 2009 study confirmed that the epicenter of *T. gondii* exposure in otters was not located within Estero Bay, as erroneously asserted by NRDC (2006) and Miller et al (2002). More importantly, Johnson et al. (2009) also hypothesized that, based in part on the new epicenter location, "*a more important source of infection might be bobcats and mountain lions*" instead of housecats. In fact, the world's largest reported outbreak of human toxoplasmosis was linked to a municipal drinking-water reservoir in British Columbia that had been contaminated by cougar feces (Bowie et al. 1997; Aramini et al. 1998).

Nevertheless, the MBCSD has maintained the public outreach and education efforts on the proper disposal of cat litter. Irrespective of the lack of a nexus with otter *T. gondii* infections, cat litter, even the "disposable" kind, can lead to sewer line blockages. As part of the outreach program, the MBCSD occasionally includes newsletters with collection-system BMPs in monthly water bills. The inserts also provide information on the newly implemented water-conservation requirements, proper disposal of pet wastes, and emergency contact information for sewer emergencies. Additionally, discussions of cat litter BMPs were included in presentations made to local realty groups as well as to members of the public during tours of the treatment plant.

¹ <http://www.cayucossd.org/>

² Frequently Asked Questions

³ <http://www.morro-bay.ca.us/342/Wastewater-Treatment-Plant-Operations>

⁴ <http://www.morro-bay.ca.us/documentcenter/view/1201>

In particular, on August 17th, John Rickenbach (WRF Deputy Program Manager), Bruce Keogh (Wastewater Division Manager), Rob Livick (Public Works Director, City of Morro Bay), and Robert Enns (President, Cayucos Sanitary District Board), gave presentations at the Scenic Coast Association of Realtors Bimonthly Education Meeting held in Morro Bay. The presentations included a summary of the progress on the water reclamation facilities currently in development by both the City and the District, and the sewer fee schedules for the City and Sanitary District. The presentations also described BMPs relating to cat-litter disposal, pharmaceuticals, and FOG, as well as information on the Household Hazardous Waste Facility located the WWTP and the lateral inspection program.

MBCSD does not anticipate any changes to the cat-litter public-outreach program in the coming year. They will continue to use newsletters, public presentations, and plant tours to communicate with the public on the topic of cat litter and its proper disposal. In addition, MBCSD staff will continue to make periodic visits to specific commercial and professional establishments to encourage them to establish and implement appropriate policies and procedures to dispose of feline wastes properly.

Other public-outreach endeavors by the MBCSD include its involvement in the collection of household hazardous wastes. Beginning in August 2000, the MBCSD collaborated with the Integrated Waste Management Authority to establish a permanent household hazardous-waste collection facility at the treatment plant. The WWTP offers free waste disposal to all residents of San Luis Obispo County every Saturday from 11:00 a.m. to 3:00 p.m., except holiday weekends. The facility remains one of the top waste-disposal sites in the county in terms of the volume of material collected. Between 20 and 50 individuals utilize the facility each weekend. From July 2015 to June 2016, the WWTP disposal facility processed nearly 20 tons of household hazardous waste, a large portion of which was recycled.

Source Identification

Past and ongoing efforts to eliminate or reduce contaminants entering the WWTP's wastestream have been successful, and as evidence of that success, elevated contaminant concentrations within effluent samples are rarely detected during the periodic chemical assays. Although nothing unusual was detected within the wastestream during 2016, anomalous concentrations of individual chemicals have been reported in the past. On those occasions, MBCSD personnel successfully traced the contaminants to the source and worked with the source owner to eliminate the contamination.

In 2006, the State Water Resources Control Board adopted new statewide waste-discharge requirements for sanitary sewer systems, which transferred responsibility for managing the introduction of FOG, and other components from the WWTP to the Collections Department under the City of Morro Bay's Sewer System Management Plan¹ and the Cayucos Sanitary District.² Ongoing source identification and resolution efforts conducted by the City and District include a grease-trap inspection program for businesses subject to the requirements. Collections personnel regularly conduct scheduled inspections as well as spot checks as described in the *Industrial Waste Survey* subsection above. During inspections, personnel discuss BMPs with restaurant staff, provided educational materials such as a BMP handbook and make recommendations for grease trap maintenance as necessary.

Known light-industrial dischargers were also inspected for compliance with source control processes and procedures. One of the most important of the Class 1 facilities is the commercial laundry that operates within the City of Morro Bay. Although this facility only contributes approximately 1.6% to plant flow,

¹ City of Morro Bay Sewer System Management Plan, originally approved June, 2009, reapproved June, 2014. City of Morro Bay Public Services Department, Wastewater Collections Division

² [CSD Ordinances.pdf](#)

commercial laundries use industrial-grade detergents, bleaches, surfactants, and brighteners that can potentially harm the bacteria within the WWTP's secondary-treatment system. In addition, solvents, oils, and other substances removed from soiled laundry have the potential to release contaminants into the wastestream. To minimize this potential, the laundry installed a pretreatment system during 2002 that included pH buffering with acid to neutralize excessive increases in wastewater alkalinity that can potentially interfere with the plant's treatment process. After a routine inspection during 2016 found the laundry's discharge ranged beyond the 5.5-to-9.0 pH allowed in the municipal code, the facility manager upgraded the laundry's pretreatment system with new chemical tanks, flash mixer, pH probe, and chemical dose pump, and changed the pH neutralization chemical from sulfuric to hydrochloric acid.

2.2 WASTEWATER CHARACTERIZATION

Monthly wastewater characterizations documented a number of different aspects of the treatment plant's performance during 2016 (Table 2.2). Removal rates quantified the plant's ability to reduce major organic constituents within the wastestream. Effluent concentrations characterized the overall quality of effluent discharged through the ocean outfall, while mass emissions quantified the cumulative load of wastewater constituents introduced into the marine environment.

Table 2.2 Monthly Averages of Influent and Effluent Parameters

Month	Flow (MGD)	Suspended Solids				Biochemical Oxygen Demand			
		Influent (mg/L)	Effluent (mg/L)	Removal (percent)	Emission (kg/day)	Influent (mg/L)	Effluent (mg/L)	Removal (percent)	Emission (kg/day)
January	0.895	214	47	— ¹	160	304	82	72.1	311
February	0.823	302	43	87.7	133	323	68	78.2	213
March	0.993	361	26	91.5	95	313	55	82.0	212
April	0.817	419	28	93.6	86	370	46	87.4	147
May	0.795	356	29	87.9	85	360	61	83.1	169
June	0.834	374	37	90.1	114	395	60	84.5	193
July	0.922	424	38	89.9	132	429	80	81.3	275
August	0.819	366	32	90.4	96	445	69	84.5	211
September	0.761	383	49	87.3	135	420	58	86.2	161
October	0.755	358	51	85.2	145	425	77	82.0	217
November	0.781	376	42	89.0	114	408	64	84.4	186
December	0.881	377	24	92.9	82	408	39	90.3	121
Average	0.840	358	37	90.0²	114	384	64	82.9	202
Monthly Limitation	≤2.36³		≤70	≥75.0	≤546		≤120	≥30.0	≤936
Annual Total (MT)					42				74
Nominal Annual (MT/year)					≤199				≤342

¹ The “dash” symbol (—) indicates that determination of the monthly TSS removal rate was not required for January 2016 because the applicability of removal rates is conditional. Specifically, the average influent TSS concentration (214 mg/L) was below the 240 mg/L threshold where enough suspended solids are present in the influent to achieve a 75% removal rate, and not result in a requirement for an unrealistically low effluent TSS concentration (<60 mg/L). Refer to the recommended clarification of this conditional removal-rate requirement in Section 5.4.

² The annual average TSS removal rate excluded inapplicable data that were collected when influent TSS concentrations dropped below 240 mg/L (refer to measurements made below the removal-rate applicability threshold shown by the grey region in Figure 2.6c).

³ Peak Seasonal Dry-Weather Flow (PSDWF)

Table 2.2 (continued) Monthly Averages of Influent and Effluent Parameters

Month	pH		Turbidity (NTU)	Settleable Solids (ml/L)	Median ¹ Total Coliform (MPN/100ml)	Oil and Grease		
	Influent	Effluent				Influent (mg/L)	Effluent (mg/L)	Emission (kg/day)
January	7.8	7.4	47	<0.1 ²	2	54	≈3.2 ^{3,4}	≈9
February	7.9	7.5	41	<0.1	4	74	6.0	19
March	8.0	7.6	34	<0.1	2	89	≈3.0	≈11
April	7.9	7.6	35	<0.1	<2	42	≈3.5	≈11
May	7.9	7.6	30	<0.1	<2	160	≈3.6	≈11
June	7.8	7.5	38	<0.1	8	90	≈2.4	≈8
July	7.7	7.5	36	<0.1	2	10	≈2.6	≈9
August	7.7	7.5	32	<0.1	<2	150	≈1.5	≈5
September	7.6	7.4	38	<0.1	<2	20	≈1.7	≈5
October	7.5	7.4	56	<0.1	<2	54	≈3.2	≈9
November	7.5	7.4	49	<0.1	<2	110	≈3.9	≈12
December	7.6	7.4	25	<0.1	<2	100	<0.8	<2.5
Average	7.7	7.5	39	<0.1	<2	79	≈3.2	≈10
Monthly Limitation	6-9	≤75	≤1.0	≤23			≤25.0	≤195

Treatment-plant personnel periodically collected wastewater samples throughout 2016. Results from the analyses of those samples were used to compute the monthly averages of the principal influent and effluent characteristics listed in Table 2.2. WWTP personnel performed the laboratory analyses to determine the principal physicochemical properties of the effluent, including concentrations of suspended solids, BOD, pH, total residual chlorine (TRC), turbidity, settleable solids, and total coliform bacteria. The frequency and the duration of individual sampling and testing events varied among the parameters. For example, average reductions in suspended solids and BOD were determined from 24-hour composite samples of influent and effluent that were collected and analyzed at least weekly. Analyses for the concentrations of most of the remaining constituents were conducted only on effluent samples. Discrete effluent grab samples were analyzed for pH, TRC, temperature, turbidity, and settleable solids on a daily basis. Effluent grab samples were also analyzed for total coliform five times per week, whereas O&G concentrations were determined by BC Laboratories from weekly samples. Effluent ammonia was determined from monthly grab samples.

Detailed analyses of these measurements confirm that, during 2016, the plant regularly exceeded wastewater treatment expectations established in regulatory standards and the original WWTP design criteria. Moreover, the plant has consistently discharged exceptionally high quality effluent over its long history, and there has never been an indication of deteriorating plant performance. Rare exceptions to standards or criteria have been brief, and have often resulted from unavoidable repairs to, or mechanical failures of a treatment-system component. However, as a byproduct of a diligent preventative maintenance program, the plant has operated at a high level of efficiency with little equipment down time.

¹ Computed from samples collected in the 30 days prior to the last day of the month (MBCSD 1997a)

² The “less-than” symbol (<) indicates that the substance was not detected at a concentration above the method detection limit (MDL), which is listed after the “<” symbol.

³ The monthly median O&G concentration is used to evaluate plant performance because it provides a more robust measure of central tendency when one or more individual concentrations are not reliably quantified (see Section IV.C.8.C of the COP). O&G concentrations measured in all but 13 of the 52 weekly effluent samples were below the 5-mg/L practical quantification limit (PQL), as shown by the gray shading in Figure 2.9c on Page 2-27.

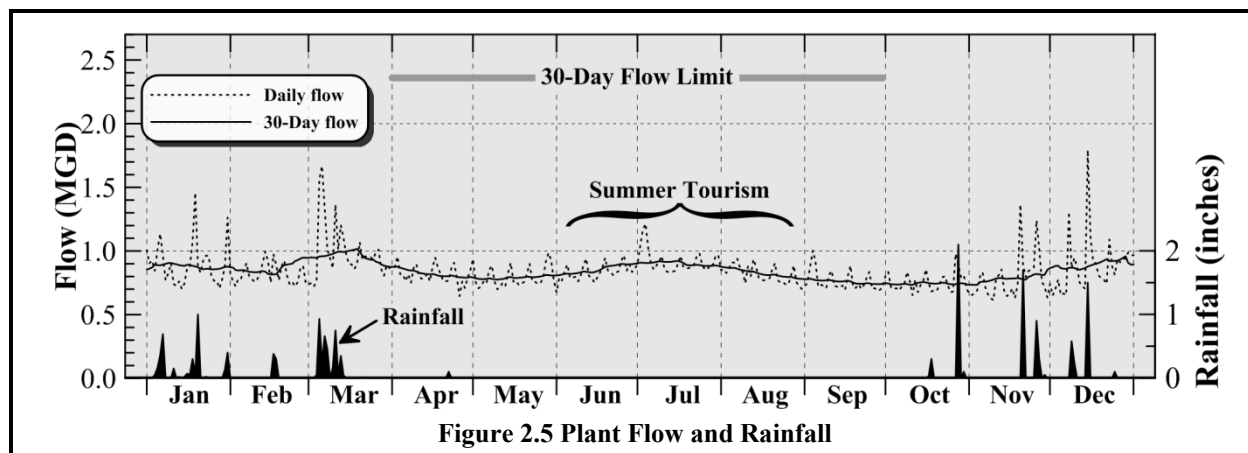
⁴ The “approximation” symbol (≈) indicates that the median concentration for the month was also too low to be reliably quantified; namely, it was below the practical quantification limit (PQL).

The primary challenge for plant personnel is to respond quickly to unanticipated failures in system components and to external events that affect the treatment process. For example, an unexpected decline in influent quality temporarily impacted treatment performance during October 2016, as described in Section 2.2.2 below.

2.2.1 Flow Rate

Flow through the plant in 2016 remained far below both the plant's design capacity and the limits established in the NPDES discharge permit. The waste discharge requirements (RWQCB-USEPA 2009) state that the "*peak seasonal dry weather flow shall not exceed a monthly average of 2.36 MGD.*" Plant throughput never approached this flow limitation during 2016 (Table 2.2), not even during winter (October through March) when the limitation does not apply because increased flow from precipitation events is expected to occur. In fact, the highest average monthly flow, 0.993 MGD in March, was only 42% of peak dry weather limit.

Additionally, this peak monthly flow occurred during the wet-weather season, when I&I associated with a series of rainstorms that occurred throughout the first half of March (solid black spikes at the bottom of Figure 2.5) demonstrably contributed to the long-term average WWTP throughput (solid black line in the Figure). This storm series brought 3.7 inches of rain to the area over a period of 11 days, and generated the second and third highest daily flows, which exceeded 1.5 MGD (dashed line in Figure 2.5).



The highest daily plant throughput (1.79 MGD) was recorded on 15 December. Although the 1.5 inches that fell on that day was not as large as the 2.1 and 1.7 inches that fell during storms in October and November, this short-lived rainwater inflow combined with flow contributions from increased groundwater infiltration into the collection system. Starting at the beginning of November, a steady infiltration-related increase is apparent in the 30-day running average flow (right-most portion of the solid line in Figure 2.5). The increased infiltration probably arose from an elevated water table that resulted from storms that began in mid-October. Overall, the 15.86 inches of precipitation recorded at the WWTP during 2016 was nearly equal to that of the prior three years combined.

Population increases related to summer tourism also resulted in an increase in long-term plant-flow (upward bow in the solid line from June through August) that was nearly comparable to that of year-end infiltration. This long-term tourism-induced flow peaked in mid-July at a flow-rate (0.92 MGD) that was only 39% of the monthly allowance for peak dry-season flow shown by the thick shaded line that spans the 'dry' season (April through September) in Figure 2.5. Short-term increases in flow due to tourism are

also apparent around the Independence Day and Labor Day holidays at the beginning of July and September. Population increases between the first and sixth of July contributed an additional 1.1 MG of wastewater to the collection system.

Although the influence of I&I and population fluctuations are apparent in the flow record throughout 2016, the total annual flow was the lowest on record. The annual average flow rate of 0.84 MGD in 2016 was far lower than any recorded in the 31-year record. It was only 90% of the flow recorded in the prior three years, which also constituted record lows compared to previous years. At least some of this decrease was related to the successful water-conservation measures implemented by the citizens of Cayucos and Morro Bay. For example, the City of Morro Bay reduced water usage in 2015 by 13.5%, significantly surpassing the 12% mandatory water restriction goal imposed by statewide limits that went into effect in April 2015.

2.2.2 Suspended Solids, Turbidity, and Settleable Solids

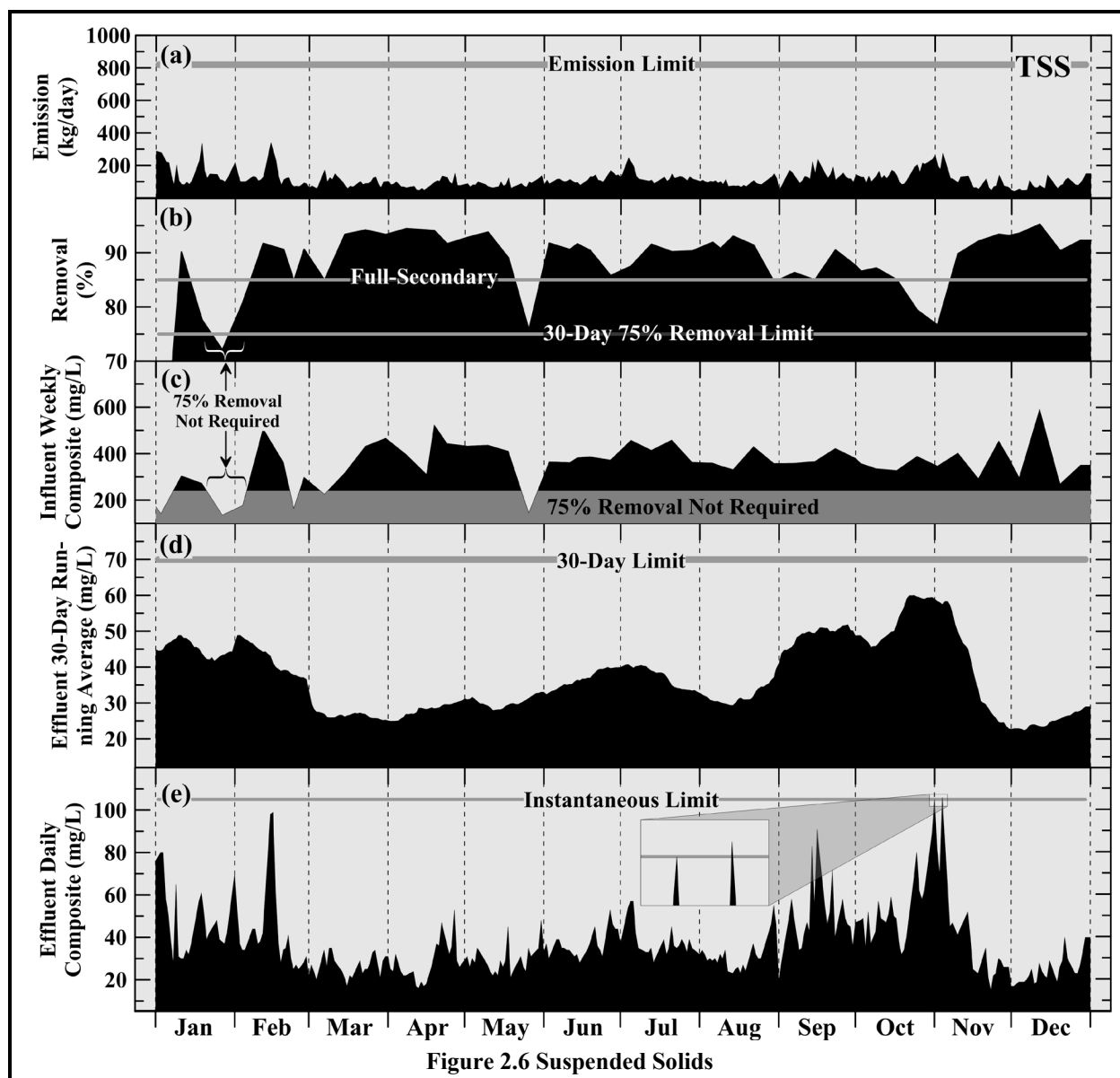
Suspended solids, turbidity, and settleable solids measure the particulate load within wastewater. One of the principal functions of the treatment process is to remove organic particulates from the wastestream. The treatment plant's removal of nearly all (90%) of the solids from influent (Figure 2.6b; Table 2.2) documents a high overall performance during 2016. In particular, the plant's solids removal rate far exceeded the minimum required by the NPDES permit, which specifies removal of only 75% of the suspended solids on a monthly basis.

In fact, the plant's 90% annual removal rate substantially exceeded the 85% monthly removal rate established for full-secondary treatment. Moreover, the WWTP achieved removal rates equivalent to full secondary treatment consistently throughout the year, with monthly averages exceeding the standard in every month except January, when the removal rate was immaterial because influent solid concentrations were already low (refer to the coinciding downward excursions in Figure 2.6bc). The removal-rate limit specified in the COP and NPDES discharge permit is a technology-based requirement rather than one that directly reflects the potential for environmental impacts within receiving waters. As such, its application is conditional on the presence of high TSS concentrations within influent, as described below.

Sharp reductions in influent TSS concentration, such as those shown in Figure 2.6c, arise from external events unrelated to the treatment process. They often occur when rainwater inflow from intense winter storms dilute the suspended solids load within the collection system. Although these low-influent TSS events are clearly unrelated to the efficacy of the treatment process, they weigh heavily in computation of removal rate. Because of this, removal rates are considered an irrelevant measure of plant performance when influent TSS concentrations are low. From a process standpoint, the same fraction of suspended solids cannot be removed from the influent stream when there is less material available to remove. From an environmental standpoint, only the solids loading within the discharge stream is of concern, not the influent loading or the fraction of material removed.

Regulators recognize that requiring a high removal rate is unnecessary when effluent TSS concentrations are low to begin with. Consequently, as discussed in Section 5.4, the NPDES discharge permit does not impose the 75% monthly removal rate when the requirement would result in an effluent TSS concentration of less than 60 mg/L (RWQCB-USEPA 2009, SWRCB 2005). Thus, the 75% removal-rate requirement does not apply when average influent TSS concentrations fall below 240 mg/L,¹ which occurred during the month of January.

¹ 240 mg/L is four-times the 60-mg/L threshold on effluent concentration that determines the applicability of removal rate for compliance evaluations.



Compared to removal rate, the overall solids load released to the ocean provides a far better measure of the discharge's potential for environmental effects. Daily mass emissions of solids were uniformly low throughout 2016 and even on maximum-emission days, they never reached even half of the allowed amount (Figure 2.6a). Low overall effluent TSS concentrations, combined with the unusually small flow rate in 2016, resulted in an annual total solids emission that was far below the allowable solids emission (Table 2.2). Over the entire year, the WWTP only discharged 42 metric tons of suspended solids to the ocean. That emission was only 21% of the projected 199 metric tons that would have been discharged if effluent had contained the permitted TSS concentration of 70 mg/L and throughput reached the nominal average dry-weather flow of 2.06 MGD.

Similarly, the long-term record of effluent TSS concentrations provides a good indication overall plant performance for any given period during 2016 (Figure 2.6d). Not only were the averages for each calendar month well below the monthly limit of 70 mg/L (Table 2.2), but the 30-day running mean never approached the limit at any time during the year (Figure 2.6d). Nevertheless, average TSS concentrations were noticeably higher for three weeks after mid-October (Figure 2.6d) because of a decline in the performance of the secondary treatment process. It led to the highest reported monthly TSS concentration (51 mg/L for October in Table 2.2), and culminated with a 106-mg/L daily TSS concentration on November 4 that slightly exceeded the 105-mg/L permitted maximum (inset in Figure 2.6e).

The decline in secondary treatment performance within the WWTP probably resulted from a combination of external factors that negatively impacted influent quality and reduced plant throughput (MBCSD 2016k). First, the odor-control compound used to limit hydrogen sulfide formation within the collection system had been changed to a nitrogen-based formulation that was later found to affect settling within the wastewater treatment processes. To correct this problem, a return to using ferrous chloride for odor control was instituted on November 11. The second contributing factor was the extraordinarily low flow into the WWTP, which further impacted the secondary process during this period. The decline in 30-day flow rate that began August 2016 (solid line in Figure 2.5) reached a record low of less than 0.74 MGD at the end of September, and remained near that level through the beginning of November, when contributions from rainwater inflow began to have an effect. This period of record-low plant throughput coincided with a sustained reduction in TSS removal efficiency (Figure 2.6b) and the attendant increase in 30-day TSS concentrations (Figure 2.6d). Changes in both of these factors led to a rapid improvement in TSS removal efficiency in mid-November (Figure 2.6b) and a sharp reduction in daily effluent TSS concentrations (Figure 2.6e).

Effluent turbidity is a related measure of solids loading. Not surprisingly then, effluent turbidity also exhibited an increase in its 7-day and 30-day running averages in conjunction with the October decline in secondary treatment performance (Table 2.2 and Figure 2.7ab). In contrast to TSS, however, none of the permitted turbidity levels was exceeded in 2016. For the most part, turbidity levels measured within individual daily grab samples were low (Figure 2.7c), and reflect levels too low to have a perceptible impact on receiving seawater clarity after discharge. Accordingly, Chapter 3 of this report presents analysis of transmissivity measurements collected offshore that demonstrate that turbidity associated with effluent discharge dissipates rapidly upon discharge. Perceptible changes in water clarity associated with effluent particulate loading are only found extremely close to discharge ports near the seafloor where little ambient light penetrates. The general absence of perceptible turbidity impacts on receiving-water clarity is visually apparent in the diffuse cloud emanating from the diffuser shown in Figure 2.2 on Page 2-5. In fact, some of the distortion apparent in the photograph is an artifact of differences in the refractive index at the seawater-wastewater interface rather than particulate loading.

Settleable solids concentrations, which are the only remaining solids-related measures of effluent quality, echo the above findings that overall particulate loads of all types were low during 2016. Specifically, the monthly averages not only remained well below the permitted limits, but were also below the 0.1-ml/L detection limit (Table 2.2). Only one of the 366 daily effluent samples contained a detectable amount of settleable solids, and at 0.4 mg/L, it too remained below the monthly average limit of 1 ml/L, as well as the weekly and instantaneous limits of 1.5 ml/L and 3.0 ml/L.

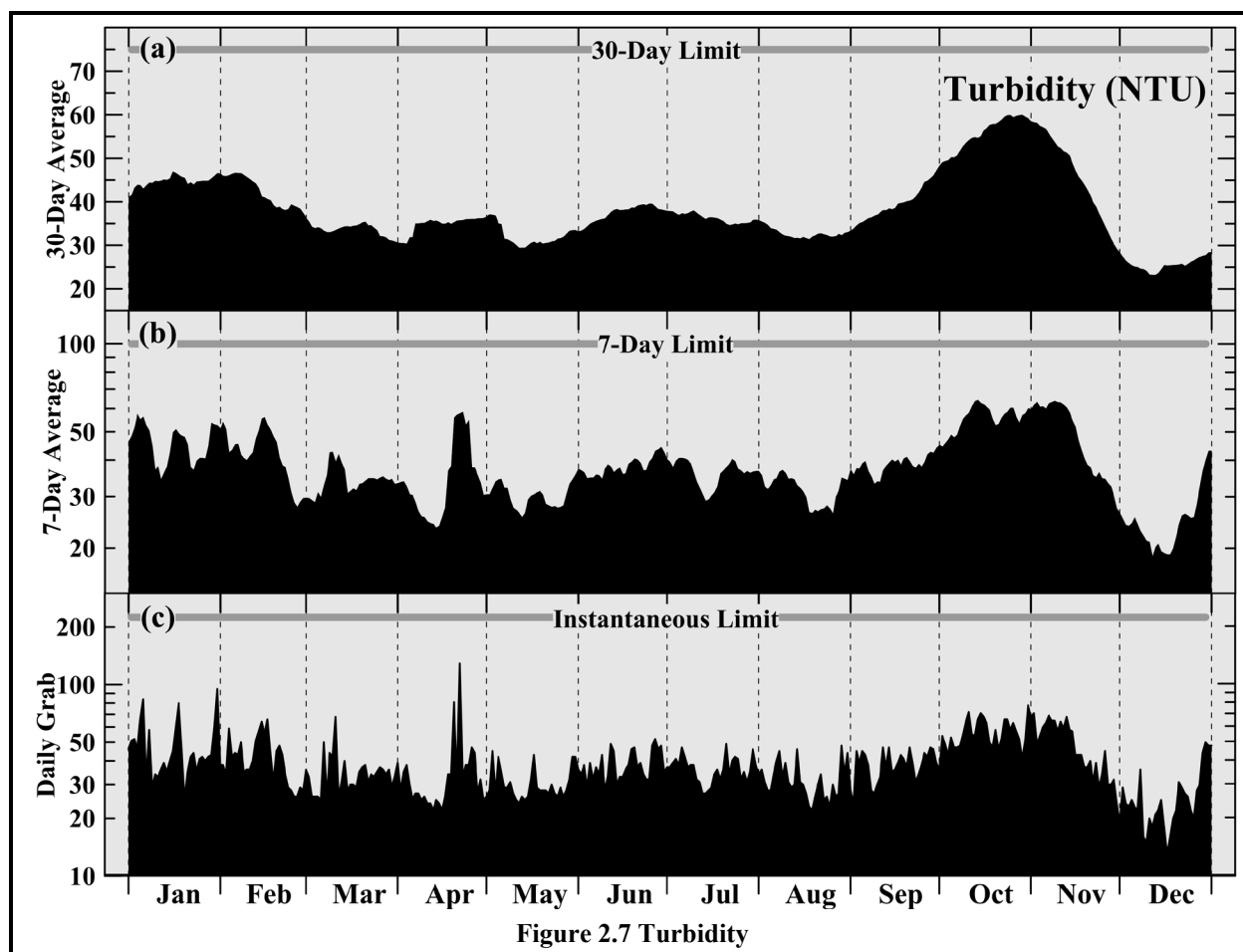


Figure 2.7 Turbidity

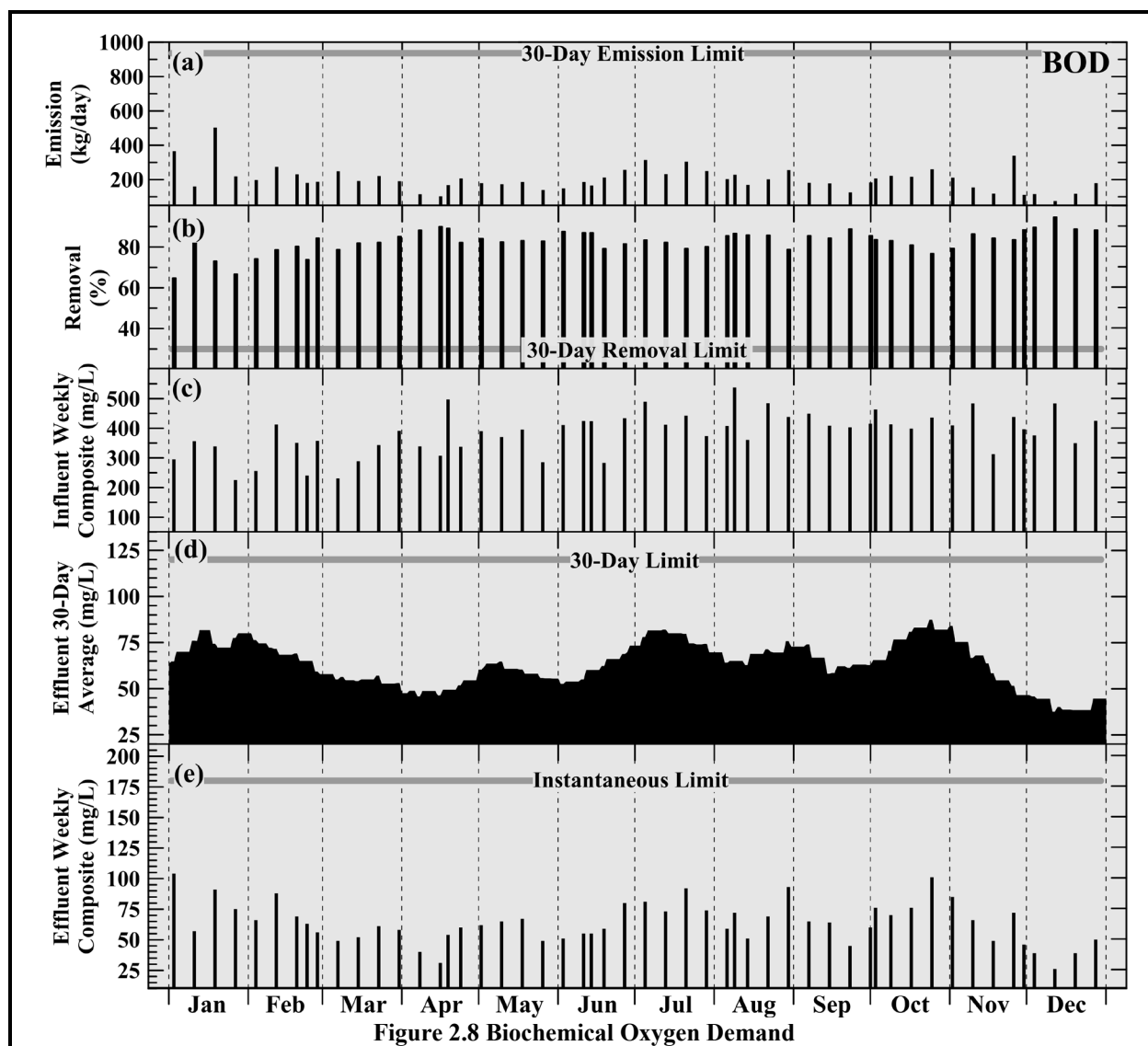
Thus, during 2016 as in prior years, all measures of effluent solids demonstrated that the treatment process exceeded performance expectations by regularly removing a greater amount of solids from the influent stream, and by discharging a small fraction of the maximum anticipated load to the marine environment. The consistently low monthly averages for effluent TSS, turbidity, and settleable solids attest to the overall effectiveness of the treatment plant's screening, grit removal, sedimentation, filtration, and clarifying processes (Table 2.2).

2.2.3 Biochemical Oxygen Demand

In combination with solids removal, a primary function of the treatment process is to reduce organic material within the wastewater stream. The effectiveness of the organic removal process is closely linked to solids removal because the majority of organic constituents are associated with wastewater particulates. However, the measure of organic loading, namely BOD, differs from the direct physicochemical measurements of turbidity and solids concentrations. Instead, BOD indirectly measures organic loading within the wastewater stream by determining the amount of oxygen required for aerobic bacteria to decompose organic matter in a sample of wastewater. Organic material, which supports bacterial degradation and demands oxygen, can harm the environment if its decomposition severely depletes DO within receiving waters. Specifically, prolonged oxygen depletion can disrupt benthic and demersal communities and can also cause mass mortalities of aquatic life (Diaz and Rosenberg, 1995).

However, DO depletion is typically only of concern in semi-enclosed water bodies, such as bays and estuaries, which are environments that fundamentally differ from the highly oxygenated open-coastal marine environment of Estero Bay. In fact, because of the ocean's higher oxygen-replenishment capabilities, an evaluation by the National Academy of Sciences (1993) questioned the environmental benefits of imposing a technology-based BOD limit on open-ocean dischargers, namely, requiring secondary-treatment standards for BOD.

Nevertheless, the NPDES discharge permit sets limits on the discharge of BOD, and independent of potential environmental influence, BOD constitutes an important measure of the overall performance of the treatment process. However, because of the complexity and duration of BOD determinations, its evaluation is only required on a weekly basis (Figure 2.8). In combination with the high overall performance of the solids removal process during 2016, BOD removal was also exceptionally high. Annually, the WWTP reduced influent organics by more than 82.9%, as determined by the average of

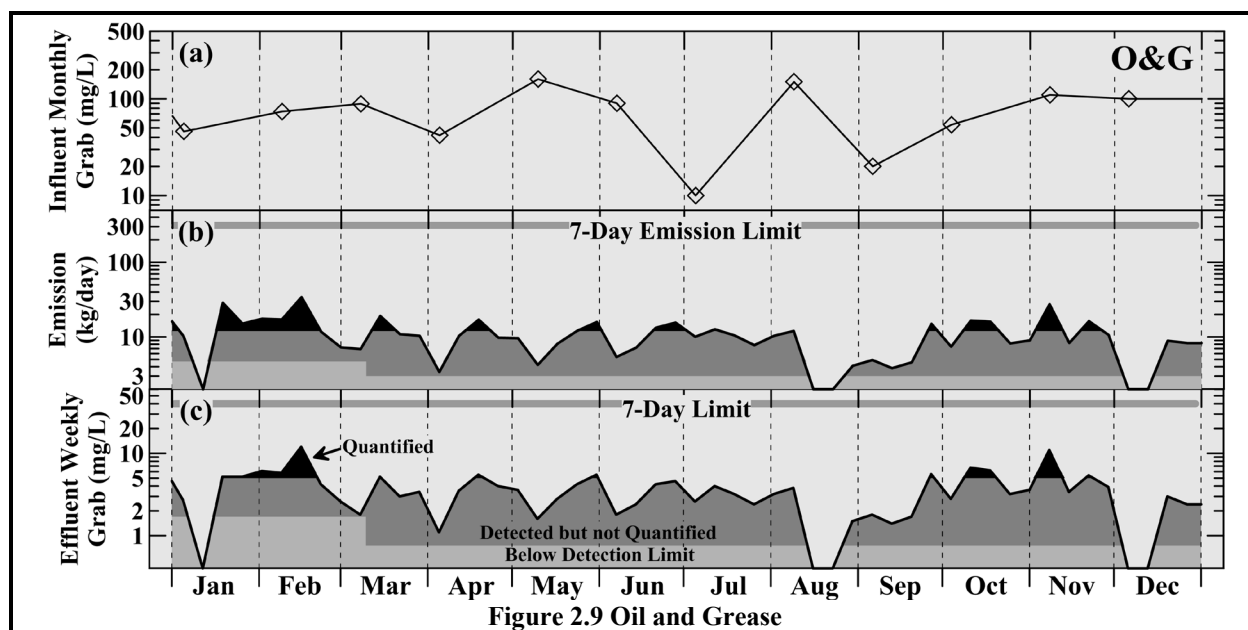


weekly composite samples analyzed for BOD (Table 2.2). Thus, the WWTP removed organic material at a rate nearly 2½ times greater than the 30% removal rate required by the NPDES discharge permit (Figure 2.8b). The monthly removal rate even exceeded the 85% removal rate established for secondary treatment in three months, and the removal approached secondary treatment in most other months by exceeding 80% all but two months.

BOD concentrations within individual effluent samples were consistently low throughout the year, and all were far below the instantaneous permit limit (Figure 2.8e). As a result, the running 30-day average remained well below the limit that applies to averages over calendar months (Figure 2.8d). This includes the October-November period when TSS removal in the secondary process was affected by a decline in the influent quality. Even so, as with TSS, the overall low BOD concentrations combined with extraordinarily low flow in 2016 resulted in a low annual BOD emission of 74 MT, which was only 22% of the 342 MT allowed by the discharge permit (Figure 2.8a).

2.2.4 Oil and Grease

During 2016, the treatment process reduced average O&G concentrations within the influent stream by at least 25-fold (Table 2.2). The actual amount of the reduction cannot be precisely determined because O&G concentrations within all but 13 of the 52 weekly effluent samples were too low to be quantified. The five quantifiable concentrations, which largely occurred in February, October and November, only marginally exceeded the 5.0-mg/L PQL specified for the O&G analysis method (black shading in Figure 2.9c). Although the presence of O&G was detected within some of the remaining 39 samples, five of the samples had concentrations too low to be detected even with the sensitive analysis method (light gray shading in the Figure).

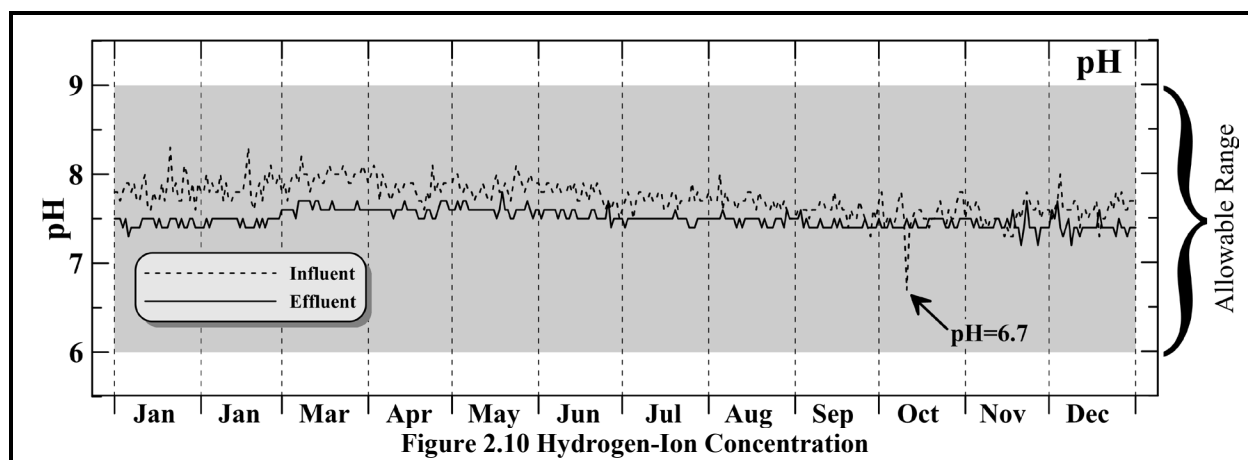


The long series of consistently low O&G concentrations measured in effluent samples collected during 2016, and in recent prior years, were unusual in the three-decade-long database. However, the recent reduction in effluent concentration did not arise because of a marked decrease in the influent concentrations (Figure 2.9a). Instead, the annual average influent concentrations in 2016 were higher than

the three-decade-long average. When combined with the lowest flow on record, the 2016 weekly emissions were an order of magnitude below the allowed emission (Figure 2.9b).

2.2.5 pH

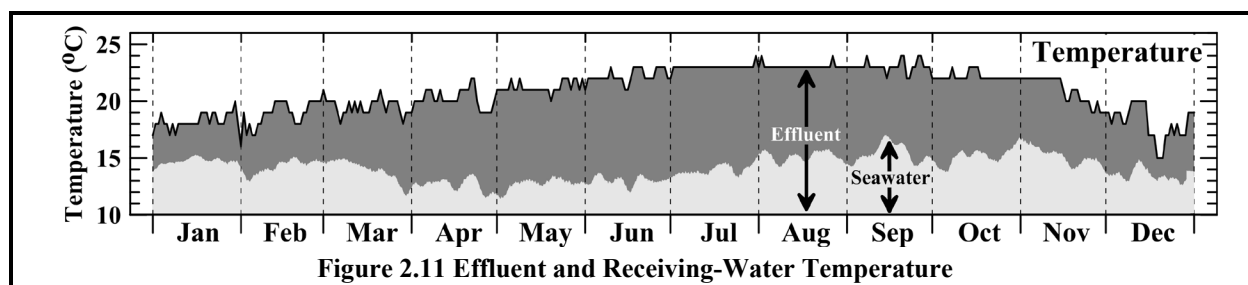
The MBCSD discharge permit requires that hydrogen-ion concentrations (pH) within effluent samples remain between 6 and 9 at all times. Although Section 301(h) of the Clean Water Act allows an NPDES permit to be issued that exceeds these pH limitations, the MBCSD discharge permit does not allow this exception. The general absence of heavy-industrial input into the collection system creates an influent stream with a nominal pH that routinely meets the discharge requirement even without treatment. Thus, because influent pH (dashed line in Figure 2.10) remained within the discharge limits (shaded area) throughout 2016, effluent pH measurements also remained within the allowable range by default (solid line). Comparison of the two time histories demonstrates that the treatment process significantly moderated short-term pH fluctuations within the influent, and slightly reduced its pH. Average annual effluent pH (7.50) was slightly less alkaline than the influent (7.73) and substantially less alkaline than the receiving seawater (8.0).



The only facility capable of discharging wastewater into the collection system with substantially different pH is a commercial laundry facility located in Morro Bay. As discussed in Section 2.1.5, their discharge is regulated under the City of Morro Bay Municipal Code that requires that the pH of the discharge remain between 5.5 and 9. The facility has an on-site acid-neutralization system to ensure compliance. Their discharge does not appear to have contributed to the unusually low influent pH measured on October 11 (Figure 2.10). WWTP personnel noted the 6.7-pH outlier during routine sample-analysis at 10:40 AM and collected a confirmatory influent sample 15 minutes later, which measured a nominal 7.7 pH. A spot-check of the laundry facility's discharge later that day confirmed it was in compliance with the pH limits of the Municipal Code.

2.2.6 Temperature

Although the NPDES permit does not limit effluent temperature, it is an important physical property to document because the difference between effluent and receiving-water temperature dictates the amount of mixing that occurs shortly after the wastewater is discharged into the ocean. The warmer the effluent compared to seawater, the greater the buoyancy of the plume and the more turbulence generated by its rise within the water column (see Section 3.2 on Plume Dispersion). Effluent temperature, shown by the upper time series in Figure 2.11, exhibits a distinct semiannual cycle that tracks seasonal insolation, in a



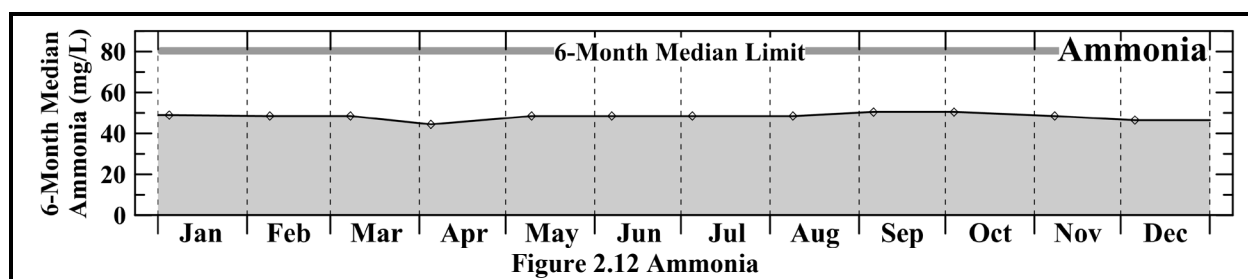
manner similar to air-temperature fluctuations. Typically, effluent temperatures begin gradually increasing in mid-winter (February), peak in mid-to-late summer (July through September), decline slightly through mid-November, and then drop relatively quickly during late fall (mid-November through mid-December).

Because of the strong and sustained influence of upwelling during 2016, seawater temperatures, shown by the time series above the light shading, did not track the seasonal-insolation trend seen in effluent temperature. In response to the onset of intense southeastward winds in early April, cool deep seawater was brought to the sea surface near the discharge location. Upwelling counteracted the warming effects of solar insolation throughout spring and into early summer (see Sections 3.2.3 and 3.2.4), and seawater temperatures were actually lower than winter temperatures during this period.

The difference in timing of insolation and upwelling produced a period of large thermal contrast, which persisted from April through September (dark shading in the Figure). Effluent temperatures were at least 6°C higher throughout most of this period. However, during June and July, the thermal contrast was much larger. It regularly exceeded 9°C as onshore surface waters continued to warm while seawater temperature was suppressed by upwelling. This large thermal contrast would normally enhance buoyancy-induced dispersion of the effluent plume significantly. However, upwelling also causes water-column stratification, which can limit vertical movement of the plume and offset the buoyancy-enhanced turbulence to some extent. The strength of upwelling winds began to decline after June and sea-surface temperatures began a slow increase that extended through October. Thus, buoyancy-induced vertical mixing of the discharge plume was greatest during the late summer and early fall of 2016 when thermal contrasts were still large and water-column stratification had decreased.

2.2.7 Ammonia

The concentration of ammonia as nitrogen ($\text{NH}_3\text{-N}$) was uniformly low throughout 2016, as has been the case in prior years. Median concentrations, computed over a 180-day period, were consistently close to 50 mg/L, which is a little more than half of the long-term permit limit (Figure 2.12). Individual grab samples analyzed for ammonia on a monthly basis during 2016 had concentrations at or below 61 mg/L, which is one-fifth of the 322-mg/L daily-maximum permit limit, and one-thirteenth of the instantaneous maximum (804 mg/L) established to protect marine aquatic life.



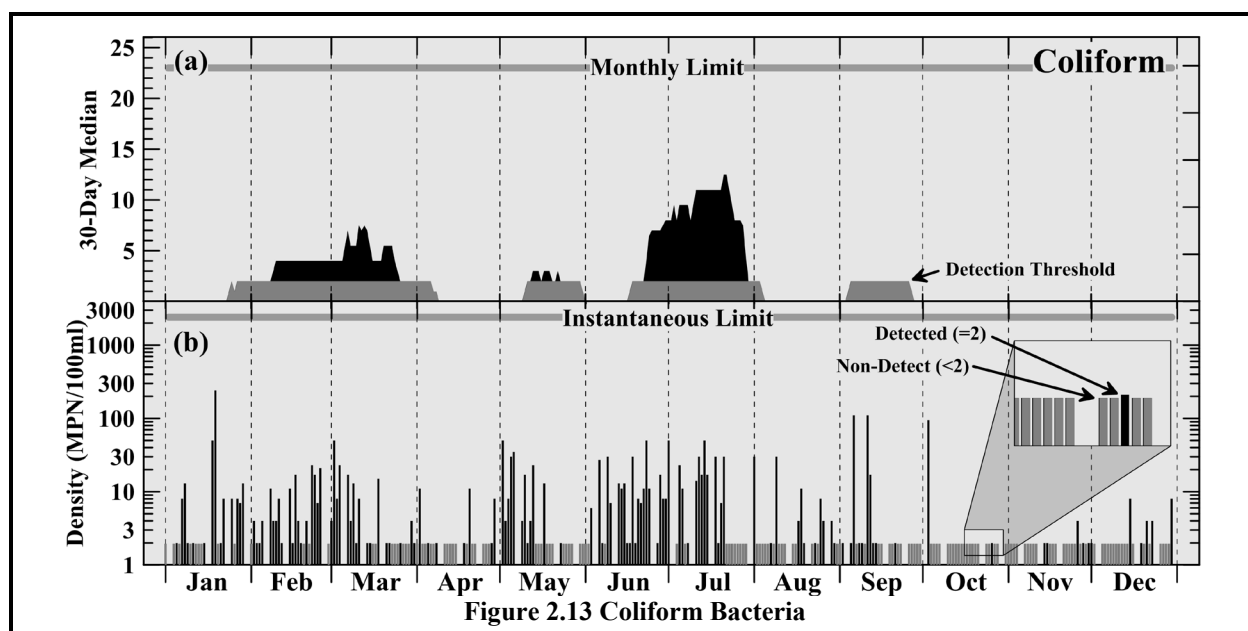
2.2.8 Residual Chlorine

Total residual chlorine (TRC) quantifies the amount of chlorine remaining in effluent grab samples that are collected after disinfection with sodium hypochlorite and subsequent dechlorination, or buffering, with sodium bisulfite. Effluent grab samples are collected when flow reaches its daily maximum, and concentrations of effluent constituents, as well as hypochlorite and bisulfite dosage, are expected to be at their highest levels. However, TRC concentrations were below the 0.05-mg/L detection limit in all 366 daily grab samples collected during 2016.

The absence of detectable TRC concentrations results from a careful balancing of the complex chlorination and dechlorination processes to obtain adequate disinfection (coliform reduction) without dosing the marine environment with excess chlorine. The complexity arises because chlorine demand is constantly changing due to continuous variations in flow and organic loading within the wastestream. Hypochlorite and bisulfite dose is normally controlled by both total-chlorine-residual and flow-paced pumps that automatically inject precisely measured amounts of these chemicals into the wastestream. However, the disinfection process occasionally requires direct operator intervention on short notice. An auto-dialing alarm system is linked to the process to notify operators by phone immediately upon an unusual excursion in TRC at any time day or night. A narrow range on the alarm points facilitate a quick response, but at the expense of numerous false alarms.

2.2.9 Coliform Bacteria

The efficacy of the disinfection process is tracked by regular measurements of coliform populations within effluent samples. As part of the NPDES permit provisions, samples collected on five consecutive days each week were analyzed for the most probable number of total coliform organisms per 100 ml (MPN/100 ml). Figure 2.13b shows that when there are detectable coliform populations, their densities vary widely among individual measurements. WWTP personnel strive to maintain densities close to the detection limit of 2 MPN/100 ml, and during 2016, 69% of the 285 measurements were at or below this detection level (see the inset in Figure 2.13b).



However, due to the complexities of the disinfection process, elevated densities are occasionally observed within individual samples. The greatest population density (240 MPN/100 ml) was measured on January 19 in conjunction with high plant flow following an intense rainstorm. Nevertheless, this isolated maximum measurement was still an order of magnitude lower than the permit limit on instantaneous coliform densities.¹

Clusters of measurable coliform density also occur when the WWTP processes an increased proportion of wastewater of sewage origin, for example, in summer months (refer to the black shading in Figure 2.13a). The longer-term trends shown in Figure 2.13a are based on a 30-day running median of the individual measurements shown in Figure 2.13b. As with the individual effluent samples, long-term coliform density remained at, or below detectable levels for most of 2016. The largest-amplitude cluster of measurable coliform density was centered within the summer tourist season, and peaked in mid-July at about half of the 23-MPN/100 ml monthly limit. However, because compliance with the monthly permit limit is based on a 30-day look-back period for each calendar month, only the February and June month-end medians of 4 and 8 MPN/100 ml ranged above the detection limit and therefore, were appropriate for comparison to the 23-MPN/100 ml permit limit (Table 2.2).

2.2.10 Toxicity

In 2016, effluent was tested semiannually, in January and July, for toxicity using chronic bioassays² of composite effluent samples (Table 2.3). The chronic bioassays found consistently low effluent toxicity, with levels much less than the discharge permit limits.

Table 2.3 Comparison of Measured Toxicity Levels with Permit Limitations

Sample Date	Bioassay Test	End Point (%)	Concentration (TU)	Limit (TU)
19 January	Red Abalone (<i>Haliotis rufescens</i>) Larval Development	5.6	17.9	134
19 July	Red Abalone (<i>Haliotis rufescens</i>) Larval Development	5.6	17.9	134

Chronic bioassays have historically been conducted using giant kelp zoospores (*Macrocystis pyrifera*). Toxicity screening studies conducted in 1993 indicated that giant kelp was more sensitive to MBCSD effluent than other species typically used in bioassays at that time, such as the larvae of the inland silverside (*Menidia beryllina*) and the bay mussel (*Mytilus edulis*) (MRS 1994). Over the following 17 years, MBCSD bioassays repeatedly demonstrated that giant kelp zoospores were only minimally affected by exposure to the treatment plant effluent. As required by the current discharge permit, however, a new screening study was conducted on effluent samples collected in July 2009 and January 2010. Those screening assays assessed the effluent's effect on the development of larval red abalone (*Haliotis rufescens*) in addition to that of giant kelp. These screening bioassays indicated that larval abalone were slightly more sensitive to MBCSD effluent than kelp zoospores; therefore, all subsequent bioassays have been conducted using larval abalone.

The chronic toxicity tests conducted in January and July 2016 measured growth response in larval red abalone after exposure to a range of effluent dilutions.³ The results of the 2016 abalone bioassays confirmed that the effluent continued to have a low chronic toxicity. All chronic-toxicity endpoints were

¹ Note the logarithmic density scale in Figure 2.13a.

² Acute bioassay testing requirements were eliminated in the current MBCSD discharge permit, in accordance with prior updates to the 2005 COP.

³ The semiannual effluent reports contain raw test data, pertinent quality assurance/quality control (QA/QC) data, and chains of custody for the chronic bioassays (MRS 2016gh).

less than one-seventh of the applicable permit limitation of 134 chronic Toxic Units (TUC) for the daily maximum toxicity. The reported TUC were based on a “*No Observable Effects Concentration*” (NOEC), which is the highest effluent concentration that does not cause an adverse effect statistically different from a control sample. They indicate that the chronic bioassays did not find adverse effects when abalone were exposed to effluent concentrations as high as 5.6%, whereas the permit allows adverse effects in concentrations as low as 0.75%.

2.2.11 Nutrients

During the review process for the current MBCSD discharge permit (USEPA 2007), concerns were raised regarding the relative contribution of nutrients discharged to the ocean by coastal treatment plants, such as the MBCSD WWTP, and their potential role in the promotion of harmful algal blooms (HABs). HABs occur when periodic explosions of growth in naturally occurring algae, which form the base of the marine food web, result in extensive monoculture blooms of particular species that are harmful to humans or other life. In addition to harm caused through the production of toxins by these species, large phytoplankton blooms can negatively affect the marine ecosystem simply from their accumulated biomass.

Considerable research has been conducted in an effort to understand the environmental factors that promote HABs. Through these studies, processes such as coastal upwelling and river runoff have been implicated as the primary factors that create physical and chemical conditions (e.g., high nutrient concentrations) conducive to the development of phytoplankton blooms (Trainer et al. 2002, Kudela et al. 2004). In particular, upwelling conditions, which are discussed in Chapter 3 of this report, have been chiefly implicated in the generation of HABs along the central California coastline (Trainer et al. 2000, Kudela et al. 2005).

In addition to these natural processes, increased human activity and pollution are also thought to be contributing factors to the recently observed increase in the frequency and intensity of HABs. For that reason, the USEPA proposed a conservation measure for “*Regular monitoring of nutrient loading from the [MBCSD] facility’s ocean outfall*” in their biological evaluation. Open-ocean dischargers, such as the MBCSD treatment plant, are not normally required to monitor for bio-stimulatory nutrients because energetic, well-flushed marine environments rapidly dilute and disperse discharged nutrients, preventing their accumulation to deleterious levels. For this reason, there are no numerical objectives for nutrient compounds (except ammonia) promulgated in the COP. However, in response to the concerns regarding nutrient loading and HABs, MRS (2008b) designed and instituted a nutrient-monitoring requirement for the MBCSD effluent monitoring program that includes the semiannual analyses of nitrate [NO_3^-], urea [$\text{CO}(\text{NH}_2)_2$], ortho-phosphate [$(\text{PO}_4)^{3-}$], and dissolved silica [SiO_2].

These particular compounds were selected because they represent limiting macronutrients for phytoplankton growth within the euphotic zone of the ocean, and have been associated with the stimulation of phytoplankton growth (Kudela and Cochlan 2000). Ammonia [NH_3] is another nitrogen compound typically associated with phytoplankton growth. However, ammonia concentrations are already regularly measured as part of the MBCSD discharge permit’s waste-discharge requirements.

In fulfillment of the current permit requirement, nutrient assays of MBCSD effluent were conducted on grab samples collected in January and July 2016. The results were consistent with those of prior years, and demonstrate that nutrient concentrations within the MBCSD effluent, and their mass loading to the marine environment from discharge, are small compared to both other central-coast dischargers and the contribution from regional streams and rivers (Table 2.4). Although concentrations of urea within MBCSD effluent (≤ 0.158 mg/L) were comparable to those of the three large central-coast WWTP’s to

Table 2.4 Nutrient Concentrations and Loading from Central-Coast Ocean Discharges

Source	Concentration (mg/L)				Mass Emission (kg)			
	Nitrate	Urea	Phosphate	Silica	Nitrate	Urea	Phosphate	Silica
MBCSD								
January	0.20	0.158	2.1	10.0				
July	<0.28	0.093	3.2	10.0	<279.	147.	3070.	12,000.
Santa Cruz	9.52	0.087	7.7	30.2	139,000.	1,360.	117,000.	489,000.
Watsonville	10.52	0.110	13.6	35.6	105,000.	1,250.	154,000.	364,000.
Monterey	4.82	0.084	3.4	41.0	85,600.	1,100.	30,300.	488,000.
Streams and Rivers ¹	3.58	0.021	0.14	25.6	1,660,000.	33,500.	340,000.	25,200,000.

the north (≤ 0.110 mg/L), the concentrations of nitrate, phosphate, and silica within MBCSD effluent were substantially lower than those of the other dischargers. The MBCSD nitrate levels, in particular, were an order of magnitude lower than those of the other WWTP's within the central-coast region. Nitrate and silica concentrations within MBCSD effluent were also less than the average concentrations found within central-coast rivers and streams; although, urea and phosphate concentrations were higher, as was the case for the other central-coast WWTP's.

Notwithstanding the slightly higher urea and phosphate concentrations within effluent, potential marine bio-stimulatory effects from nutrient discharge are dictated by the total mass emissions contributed by the various sources (right side of Table 2.4). After accounting for the relatively small MBCSD discharge, its total nutrient loading² to the marine environment during 2016 was 39-times smaller than any of the three large WWTP's, all of which discharge into the waters of the Monterey Bay National Marine Sanctuary. Similarly, total nutrient loading from the MBCSD discharge was three orders-of-magnitude smaller than the contribution from runoff within the central-coast region.

2.2.12 Chemical Compounds

In addition to the effluent properties and bioassay results described above, 78 chemical contaminants are regulated by the COP, and have their effluent concentrations limited in the discharge permit. Effluent composite samples were analyzed in January and July 2016 for the presence of these chemical compounds, which include trace metals, chlorinated and nonchlorinated phenolic compounds, volatile organic compounds, organochlorine pesticides, PCBs, cyanide, base-neutral compounds, and radionuclides (Table 2.5). The COP regulates the discharge of these compounds for the protection of marine life and the protection of human health from exposure to both carcinogenic and noncarcinogenic substances.

As has been the case in the historical record of 3,597 chemical assays spanning 24 years of MBCSD effluent monitoring, the 2016 analyses detected concentrations of only a few ubiquitous compounds at levels well below the regulatory limits. Those compounds are discussed below, while detailed discussions of effluent chemistry, along with the corresponding concentration limits, minimum reporting levels, laboratory data sheets, pertinent QA/QC data, and chains of custody for all the chemical constituents were provided in the semiannual self-monitoring reports (MRS 2016gh).

¹ Average concentrations and total emissions from fourteen streams and rivers discharging to the northern central coast from July 2005 and June 2006 (CClean 2007)

² Sum of the emissions from the four nutrient compounds listed in the Table.

Table 2.5 Chemical Compounds Detected within Effluent Samples

Compound	Concentration (µg/L)			Mass Emission (kg/vr)	
	Limit	January	July	Goal	Measured
Protection of Marine Aquatic Life					
Arsenic	670.	2.0	≈1.1 ²	17.	≈1.81
Chromium VI ³	270.	<1.2 ⁴	≈1.3	93.	<1.45
Copper	140.	20.	22.	690.	24.4
Lead	270.	0.89 ⁵	1.4	465.	1.33
Mercury	5.29	≈0.072	<0.030	1.4	<0.06
Nickel	670.	≈4.8	≈4.8	142.	≈5.59
Selenium	2,010.	4.2	≈1.5	65.	≈3.35
Zinc	1,620.	61.	70.	244.	76.
Nonchlorinated phenolics	4,020.	— ⁶	≈6.8	—	—
Radionuclides (pCi/L) α	15.	—	0.887	—	—
Radionuclides (pCi/L) β	50.	—	22.	—	—
Protection of Human Health (Non-Carcinogens)					
Chromium III ²	25,500,000.	—	≈1.3	—	—
Toluene	11,400,000.	—	≈0.23	4.	≈0.27
Protection of Human Health (Carcinogens)					
Chloroform	17,400.	—	≈0.80 ⁷	5.	≈0.93
Dioxin (pg/L)	0.52	—	≈0.0674	1.48 mg	≈0.078 mg

The chemical assays found only six of the 78 regulated chemical compounds present in quantifiable amounts within the 2016 effluent samples.¹ The measured concentrations for all six compounds were significantly less than the permitted limits. Annual mass emissions of these compounds also met the goals identified in the discharge permit's reporting provisions. The compounds with quantifiable concentrations included three trace metals (copper, lead, and zinc), arsenic, selenium, and radionuclides.

The analyses also detected the presence of seven additional compounds in the semiannual effluent samples, but at concentrations that were too low for reliable quantification. Specifically, the concentrations of these compounds were higher than the method detection limit (MDL)⁸ but less than the minimum level (ML). Reporting of these detected-but-not-quantified concentrations is required under the

¹ Quantifiable concentrations are listed in bold typeface in Table 2.5.

² The “approximation” symbol (≈) indicates that the detected concentration, designated with an “as estimated” qualifier, was too low to be reliably quantified, namely, it was below the prescribed Minimum concentration Level (ML). Accurately quantified “as measured” concentrations are indicated by bold typeface.

³ The total chromium concentration is reported instead of the trivalent or hexavalent oxidation states alone.

⁴ The “less-than” symbol (<) indicates that the substance was not detected at a concentration above the MDL, which is listed after the “<” symbol.

⁵ The reported concentration was below the practical quantification limit (PQL) and was identified “as estimated” by the chemistry laboratory. However, in accordance with the guidance from the COP and the NPDES permit, the reported value is listed “as measured” herein, because the value exceeded the 0.5-µg/L lead ML.

⁶ The “dash” symbol (—) indicates that analysis of the compound was not required as part of the monitoring program, or that a mass-emission goal was not specified in the discharge permit.

⁷ The reported concentration was above the PQL and accordingly, was not identified “as estimated” by the chemistry laboratory. However, in accordance with the guidance from the COP and the NPDES permit, the reported value is listed here as an estimated concentration because the measured value was below the 2.0-µg/L chloroform ML.

⁸ The method detection limit is the lowest concentration that can be reported under ideal conditions, when the sample contains only the compound of interest in a concentration within an optimal calibration range and within a medium that does not interfere with the performance of the analytical instrument.

current NPDES discharge permit; although they are not compared to effluent limitations for compliance determinations.

Trace Metals

Three of the quantifiable concentrations were associated with commonly occurring metals: copper, lead, and zinc. Unlike synthetic organic compounds, trace metals occur naturally within the mineralogy of sediments along the central California coast and are ubiquitous in the local sedimentary environment. All three of the metals were detected in quantifiable amounts within seafloor sediment samples collected at all the benthic survey stations, including the distant Reference Station B1 (see Section 4.2). These metals enter the wastewater collection system through erosion of natural mineral deposits along the central California coast as well as through corrosion of household plumbing systems. Regardless of their source, the effluent metal concentrations were at most one-sixth of levels deemed deleterious to marine organisms, and the highest metal mass emission was less than one-third of the emission goal recommended in the discharge permit.

These three metals have been detected at quantifiable levels in more than half of the effluent samples collected during the last 24 years; therefore, they do not represent a new or increased source of contaminants entering the collection system. Moreover, statistical analyses of the long monitoring history unequivocally demonstrates that there is no reasonable potential for these effluent metal concentrations to exceed the discharge permit limits in the future (Refer to the recommendation concerning reductions in monitoring frequency in Section 5.4).

Arsenic and Selenium

Arsenic and selenium, two naturally occurring metalloids within the mineralogy of the central California coast, were also detected within the 2016 effluent samples. Arsenic and selenium concentrations have been found to increase slightly within MBCSD effluent samples when these naturally occurring compounds are mobilized within surficial soils following rainstorms. Rainwater inflow into the collection system from the storms that preceded the effluent sample collection on 12 January (refer to Figure 2.5) may account for the presence of quantifiable concentrations found in that sample. Conversely, the absence of measureable rainfall prior to the July sampling event resulted in concentrations too low to be quantified. Regardless of their source, the amounts found in the 2016 effluent samples were two orders of magnitude below levels deemed deleterious to marine organisms, which are encapsulated in their respective permit limits.

Radionuclides

As with the trace metals and metalloids described above, the presence of measurable alpha (α) and beta (β) radioactivity within the 2016 effluent sample was common. Radionuclides have regularly been detected within MBCSD effluent samples, largely because naturally occurring radioactive material (NORM) is present throughout the earth's crust and because radioactive decay can be quantified at extraordinarily low levels. Consequently, the permit limits specified for radionuclides are not derived in the same manner as for other chemical constituents.

Permit limits for other constituents are based on their potential for adverse impacts after discharge. As such, they incorporate an allowance for the 133-fold minimum dilution that occurs immediately after discharge. Limits on radioactivity, however, are based on the California Code of Regulations, Title 22, §64441 and §64443, and are the same as those established for drinking water; they do not account for post-discharge dilution. Regardless of their origin, the levels of radioactivity measured within the July 2016 effluent sample were well within drinking-water standards.

The low decay levels measured within the July 2016 effluent samples were typical of most prior effluent samples and of naturally occurring sediments of the region. Alpha particle activity arises from natural mineral deposits that enter the collection system through erosion. Beta particle activity arises from radioactive decay in both natural and man-made materials.

2.3 BIOSOLIDS

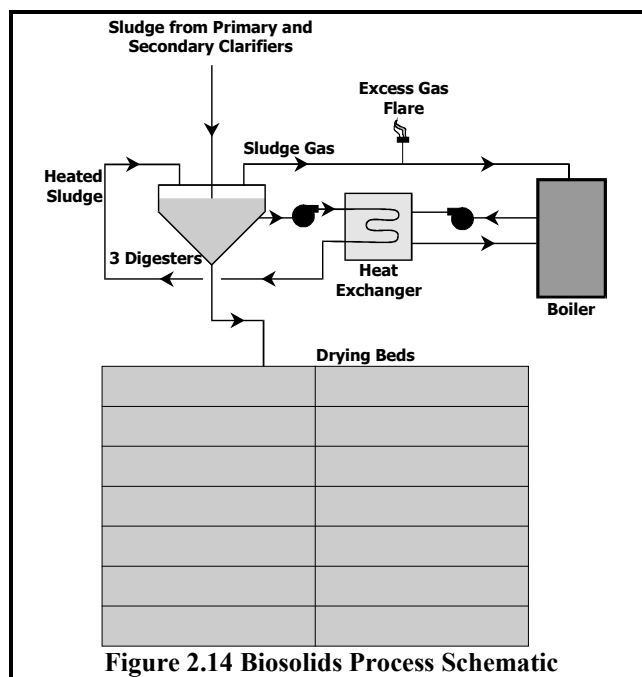
The monitoring and reporting requirements of the NPDES permit (RWQCB-USEPA 2009) stipulate characterization of biosolids in accordance with 40 CFR 503 (USGPO 1997b). To that end, a complete biosolids report covering 2016 was submitted by the MBCSD to the USEPA, RWQCB, and SLO EHS.¹ That report is incorporated in this annual monitoring report by reference, and the disposition of the approximately 246 dry metric tons of biosolids generated by the WWTP during 2016 is briefly summarized in this Section. This Section also discusses the major chemical compounds that were found within the biosolids produced by the plant, because they lend insight into the performance of the WWTP and because they determine the suitability of biosolids for future composting and land application.

2.3.1 Solids Treatment Process

Solids removed by the clarifiers (Figure 2.3 on Page 2-14) are processed as shown in the schematic of Figure 2.14. Sludge is stabilized within two mixed-primary digesters that heat the sludge to temperatures between 96°F and 98°F (36°C to 37°C). Heated sludge is transferred to a secondary digester with heat exchange from the primary digesters. Solids settle in the secondary digester and the supernate is returned to the wastewater treatment process. The sludge is transferred to drying beds. The primary digesters' capacities are 170,544 gallons and 191,500 gallons, and the secondary digester's capacity is 166,056 gallons, giving a total capacity of 528,100 gallons (2 megaL).

During 2016, solids were stabilized by pumping raw sludge to a primary digester at rate of 9,942 gallons per day. There, the average detention time was 19.3 days before the solids were transferred to the other primary digester that was combined in series with the secondary digester. Average detention time for the combined primary and secondary digestion processes was 36.4 days.

Stabilized sludge drawn from the secondary digester was transferred to one of 12 sludge-drying beds. Each of these 5,200 ft² (483 m²) beds has an under-drain and decanting system that recirculates runoff through the treatment process. Once dried, the biosolids were removed from the beds and stored in a concrete containment area that also drained rainfall runoff through the treatment system. Biosolids were



¹ Morro Bay/Cayucos Wastewater Treatment Plant 2016 Annual Sewage Sludge Report from J. Gunderlock, Wastewater/Collections Supervisor, MBCSD, to L Fondahl, USEPA Region 9, dated 17 February 2017.

stored in this area until they were removed from the WWTP. Biosolids storage times are generally less than one year.

On October 21, 25, and 26, 195 tons of biosolids were hauled to the Liberty Composting Facility, which operates under Solid Waste Information System Permit No. 15-AA-0287. The biosolids transferred to the Liberty Composting Facility were used for soil amendment after composting was completed at the facility. Prior to shipping, the MBCSD provided a Title 22 Certification for Non-hazardous Materials and a Class-B biosolid certification statement based in part on chemical analyses described below.

2.3.2 Chemical Compounds

In compliance with the Monitoring and Reporting Program, chemical analyses were conducted on a composite of biosolid samples collected from the drying beds on 10 August 2016. Those beds contained biosolids that were ready for shipment from the WWTP at the time. The full laboratory results, including chains of custody, instrument calibration reports, and analyses of method blanks and spikes, were reported by MRS (2016b). They are compared with regulatory limits in Table 2.6 on the following page.

The data in the table shows that biosolid contaminant concentrations were significantly less than regulatory thresholds that would designate them as hazardous, or that would limit their use for land application or composting. The analyses tested for the presence of more than 150 potential contaminants and measured seven other properties and nutrients within the biosolid sample. Nevertheless, only a few compounds were detected, primarily naturally occurring trace metals.

All trace-metal concentrations were below the total threshold limit concentrations (TTLC) that would designate the biosolids as hazardous. Dry-weight concentrations for all detected metals in the biosolid sample were well below the federally-mandated thresholds, including the monthly limit for materials suitable for agricultural land application (as represented in the three right-most columns of Table 2.6). The other compounds listed in Table 2.6 further characterize the biosolids, as mandated in the waste discharge requirements. Additionally, a modified WET test (STLC) for Hexavalent chromium and total dissolved solids was conducted in response to a request from the composter.

Table 2.6. Comparison between Measured Biosolid Concentrations and State and Federal Limits

Constituent	Units	Wet Weight				Dry Weight		
		Measured		Limit		Measured	Limit	
		Bulk	WET ¹	STLC ²	TTLC ³	Bulk	Monthly ⁴	Ceiling ⁵
Solids	%	86.3	— ⁶	—	—	—	—	—
Cyanide	ppm	2.6	—	—	—	3.1	—	—
Antimony	ppm	≈1.6 ⁷	—	15.	500.	≈1.9	—	—
Arsenic	ppm	1.5	—	5.	500.	1.8	41.	75.
Barium	ppm	210.	—	100.	10,000.	240.	—	—
Beryllium	ppm	≈0.083	—	0.75	75.	≈0.096	—	—
Boron	ppm	8.7	—	—	—	10.	—	—
Cadmium	ppm	0.91	—	1.	100.	1.1	39.	85.
Chromium (Total)	ppm	18.	—	560.	2,500.	20.	1,200.	3,000.
Chromium (Hexavalent)	ppm	≈0.88	ND	5.	500.	≈1.0	—	—
Cobalt	ppm	≈2.4	—	80.	8,000.	≈2.8	—	—
Copper	ppm	180.	—	25.	2,500.	210.	1,500.	4,300.
Lead	ppm	14.	—	5.	1,000.	16.	300.	840.
Mercury	ppm	0.65	—	0.2	20.	0.76	17.	57.
Molybdenum	ppm	8.3	—	350.	3,500.	9.6	18.	75.
Nickel	ppm	15.	—	20.	2,000.	18.	420.	420.
Selenium	ppm	4.7	—	1.	100.	5.4	36.	100.
Silver	ppm	1.3	—	5.	500.	1.5	—	—
Thallium	ppm	ND	—	7.	700.	ND	—	—
Vanadium	ppm	14.	—	24.	2,400.	16.	—	—
Zinc	ppm	420.	—	250.	5,000.	480.	2,800.	7,500.
Hydrogen-Ion	pH	6.61	—	—	—	—	—	—
Phosphate	ppm	39,000.	—	—	—	45,000.	—	—
Ammonia	ppm	4,300.	—	—	—	4,900.	—	—
TKN	ppm	14,000.	—	—	—	17,000.	—	—
Organic Nitrogen	ppm	9,700.	—	—	—	12,000.	—	—
Nitrate as NO ₃	ppm	1,600.	—	—	—	1,900.	—	—
Oil & Grease	ppm	27,000.	—	—	—	31,000.	—	—
Total Dissolve Solids	ppm	—	3,700.	—	—	—	—	—

¹ Waste Extraction Tests (WET) measure the soluble leachate or the extractable amount of a substance contained within a bulk sample of biosolids. A WET is indicated if the bulk wet-weight concentration of a contaminant exceeds 10 times the STLC.

² Soluble Threshold Limit Concentrations (STLC) apply to the measured concentration in the liquid extract from a biosolid sample, as determined by a WET. Biosolids with leachate concentrations exceeding the STLC are classified as hazardous in the State of California, as described in the California Code of Regulations (CCR 2003).

³ Total Threshold Limit Concentrations (TTLC) apply to the total wet-weight concentration of a contaminant within a bulk biosolid sample consisting of the entire millable solid matrix, rather than just the leachate. Biosolids are designated as hazardous wastes in the State of California if measured bulk concentrations exceed the TTLC, as described in the CCR (2003).

⁴ Federally mandated dry-weight limits imposed on biosolids suitable for application on agricultural land apply to monthly average concentrations as defined in Table 3 of the Code of Federal Regulations (USGPO 1997b). [40 CFR §503.13(b)(1)].

⁵ Federally mandated dry-weight ceiling concentrations above which biosolids are considered hazardous waste as defined in Table 1 USGPO (1997b).

⁶ “—” indicates that the measurement was not required or its limit was not specified.

⁷ Concentrations preceded by an “approximation” symbol (≈) were too low to be reliably quantified and represent estimated concentrations because they were reported below the minimum level (ML) but above the method detection limit (MDL).

3.0 RECEIVING WATER QUALITY

This chapter describes environmental conditions within the ocean waters immediately adjacent to the MBCSD wastewater discharge during 2016. The first section details the monitoring program that assesses compliance with the water-quality objectives of the California Ocean Plan (COP; SWRCB 2005) as embodied in the NPDES permit covering the MBCSD discharge (hereinafter referred to as “*the NPDES permit*,” RWQCB–USEPA 2009). The second section discusses the disposition of the effluent plume within the receiving waters. It addresses how plume dispersion is influenced by the hydraulic design of the outfall and by the stratification of the receiving waters. The final section of this chapter provides a synopsis of the offshore measurements collected during the four water column surveys conducted during 2016 (MRS 2016cdef) and compares them with the waste discharge requirements specified in the NPDES permit. Quantitative analyses of continuous instrumental measurements and the qualitative visual observations are also addressed in the last section. The observations demonstrate that the diffuser structure was operating as designed by efficiently diluting treated wastewater within the zone of initial dilution (ZID) that surrounds the outfall.

During all four of the 2016 offshore receiving-water surveys, dilute wastewater was detected within the receiving waters. The weak discharge-related perturbations in the water-property fields indicated that the treated wastewater was rapidly mixed upon discharge, and that the observed perturbations were small compared to the natural variability in seawater properties observed along this portion of the central California coast. Although excursions in transmissivity ranged beyond limits expected for receiving seawater at the time of the March and October surveys, they resulted from entrainment of turbid ambient seawater that was present in a thin layer overlying the seafloor. Based on the high initial-dilution levels measured during the surveys, maximum end-of-pipe contaminant concentrations allowed by the NPDES permit would be expected to achieve the receiving-water objectives of the COP with a large margin of safety. In summary, the highly localized, transient seawater perturbations associated with the effluent discharge were too small to be of environmental significance.

3.1 MONITORING PROGRAM

Wastewater chemistry (Chapter 2), receiving waters (this chapter), and the biology and physicochemistry of seafloor sediments (Chapter 4) constitute the three major components of the MBCSD monitoring program. During 2016, and for the past three decades, the receiving-water monitoring program has included requirements for quarterly offshore surveys conducted near the wastewater discharge within northern Estero Bay, and surfzone coliform sampling along the adjacent shoreline. Measurements collected during the offshore surveys include instrumentally recorded seawater-properties, flow velocity determined from drifter trajectories, qualitative observations of general oceanographic and meteorological conditions, and visual observations of beneficial uses and potential aesthetic impacts.

3.1.1 Objectives of Sampling

Receiving-water monitoring is required under provisions of a 301(h)-modified NPDES permit to determine compliance with the water-quality objectives of the COP (SWRCB 2005) and the Central Coast Basin Plan (RWQCB 1994). The monitoring program also satisfies the provisions of the Clean Water Act (40 CFR 125.63c), which require a receiving-water monitoring program that “*provide[s] adequate data for evaluating compliance with water quality standards or water quality criteria.*” In addition, water column measurements provide background information on the vertical structure of ambient marine waters

near the discharge location. This information aids in estimating minimum initial dilution rates, which occur when the receiving waters are highly stratified, as described in Section 3.2.2.

3.1.2 Scope of Monitoring

One goal of the monitoring program is to assess whether the discharge meets the water-quality objectives embodied in the COP. These objectives, summarized in Table 3.1 on the following page, ensure the reasonable protection of beneficial uses and prevention of nuisance. They apply to samples collected within the discharge's wastefield after initial dilution has been completed. Initial dilution is the rapid, turbulent mixing of wastewater and seawater in the area immediately surrounding the point of discharge.

Ambient seawater is forcibly entrained into the discharge plume by two physical processes. Close to the diffuser ports, the momentum of the wastewater jet induces turbulent shear that rapidly entrains near-bottom seawater, thereby diluting the effluent. Additional turbulent mixing occurs as the warm, buoyant effluent-seawater mixture rises through the water column. This buoyancy-induced entrainment can cause the density of the effluent plume to approach that of ambient seawater (neutral buoyancy) at some point in the water column. Depending on the degree of water-column stratification at the time of discharge, the plume either can reach the surface or can become trapped at some intermediate depth level. Thus, initial dilution, where a majority of effluent dilution is achieved shortly after discharge, is complete when the diluted wastewater ceases to rise in the water column and begins to spread horizontally.

To better assess compliance with the COP standards, the receiving-water monitoring program was extensively modified when a previous NPDES permit was issued in 1999 (RWQCB–USEPA 1998b). Prior to 1999, the effluent plume was rarely observed in the instrumental measurements and when it was, its lateral extent was largely indeterminate. This was partially due to the widely spaced water-sampling stations that, at the time, coincided with the benthic sampling pattern. Additionally, the earlier NPDES permit required collection of discrete water samples to be analyzed for suspended solids, coliform, and oil and grease. This water-bottle sampling was time-consuming and compromised the synoptic nature of the instrumental measurements. Beginning with the 1999 surveys, bottle casts were eliminated and the lateral extent of the survey was reduced from 1000 m to 100 m, while the number of sampling stations was doubled from eight to sixteen. Rapid instrumental sampling within this tighter sampling pattern consistently provided high-resolution synoptic snapshots of the ocean waters immediately surrounding the outfall. As a result, the disposition of the effluent plume has been accurately delineated in all 72 surveys conducted since the beginning of 1999.

Further improvements to the receiving-water monitoring program were implemented in 2009 as part of the current NPDES permit (RWQCB–USEPA 2009). For example, the navigational accuracy of positioning systems had advanced to the point where the location of individual measurements could be precisely determined relative to the discharge point. This navigational precision permitted accurate delineation of the plume's spatial extent, which is important for assessing compliance with water-quality objectives because they only apply to that portion of the plume that lies beyond the ZID. With that in mind, the receiving-water monitoring program in the current NPDES permit added a requirement for a tow survey, where the instrumentation package is towed horizontally across and around the diffuser structure at two different depth levels. Collection and analysis of this tow data began with the second-quarter receiving-water survey conducted in 2009. Additionally, all surveys since that time have included vertical profiles of seawater properties at six sampling stations aligned along a north-south transect, and spanning an alongshore-distance of 100 m on either side of the diffuser structure.

Table 3.1 Water-Quality Objectives of the California Ocean Plan

B. Bacterial Characteristics

1. Water-Contact Standards

Within a zone bounded by the shoreline and a distance of 1,000 feet from the shoreline or the 30-foot depth contour, whichever is further [sic] from the shoreline and in areas outside this zone used for water contact sports, as determined by the Regional Board, but including all kelp beds, the following bacterial objectives shall be maintained throughout the water column:

- a. The 30-day geometric mean of the five most-recent seawater samples from each site shall not exceed a total coliform density of 1,000 per 100 ml; a fecal coliform density of 200 per 100 ml; or an enterococcus density of 35 per 100 ml.
- b. A single seawater sample shall not exceed a total coliform density of 10,000 per 100 ml; a fecal coliform density of 400 per 100 ml; an enterococcus density of 104 per 100 ml, or a total coliform density of 1,000 per 100 ml when the fecal to total coliform ratio exceeds 0.1.

The "Initial Dilution Zone" of wastewater outfalls shall be excluded from designation as "kelp beds" for purposes of bacterial standards and Regional Boards should recommend extension of such exclusion zone where warranted to the State Board. Adventitious assemblages of kelp plants on waste discharge structures (e.g., outfall pipes and diffusers) do not constitute kelp beds for purposes of bacterial standards.

2. Shellfish Harvesting Standards

Throughout the water column in all areas where shellfish may be harvested for human consumption, as determined by the Regional Board, the median total coliform density shall not exceed 70 per 100 ml and not more than 10 percent of the samples shall exceed 230 per 100 ml.

C. Physical Characteristics

1. Floating particulates and grease and oil shall not be visible.
2. The discharge of the waste shall not cause aesthetically undesirable discoloration of the ocean surface.
3. Natural light shall not be significantly reduced at any point outside the initial dilution zone as a result of the discharge of waste.
4. The rate of deposition of inert solids and the characteristics of inert solids in ocean sediments shall not be changed such that benthic communities are degraded.

D. Chemical Characteristics

1. The dissolved oxygen concentration shall not at any time be depressed more than 10 percent from which occurs naturally, as a result of the discharge of oxygen demanding waste materials.
2. The pH shall not be changed at any time more than 0.2 units from that which occurs naturally.
3. The dissolved sulfide concentration of waters in and near sediments shall not be significantly increased above that present under natural conditions.
4. The concentration of substances set forth in Chapter II, Table B in marine sediments shall not be increased to levels which would degrade indigenous biota.
5. The concentration of organic materials in marine sediments shall not be increased to levels which would degrade marine life.
6. Nutrient materials shall not cause objectionable aquatic growths or degrade indigenous biota.
7. Numerical water quality objectives in Table B apply to all discharges within the jurisdiction of this plan.

E. Biological Characteristics

1. Marine communities, including vertebrate, invertebrate and plant species, shall not be degraded.
2. The natural taste, odor and color of fish, shellfish, or other marine resources used for human consumption shall not be altered.
3. The concentration of organic materials in fish, shellfish or other marine resources used for human consumption shall not be bioaccumulated to levels that are harmful to human health.

F. Radioactivity

1. Discharge of radioactive waste shall not degrade marine life.

An automated instrument package, commonly referred to as a CTD,¹ was used to collect precise *in situ* measurements of seawater properties during each of the four quarterly receiving-water surveys of 2016. Measured seawater parameters consisted of temperature, salinity, dissolved oxygen (DO),

¹ Conductivity, temperature, and depth (CTD)

acidity/alkalinity (pH), and light transmittance (transmissivity or water clarity). Qualitative visual observations made during the offshore surveys provided ancillary information on any potential wastewater contributions to levels of floating particulates, seawater discoloration, odors, algal blooms, and surface water clarity. Wildlife and recreational use, as well as general weather and sea state, were also noted.

In addition to the quarterly sampling used to delineate the discharge plume offshore, regular monitoring of the adjacent surfzone along Atascadero State Beach has historically been conducted to assess aesthetic conditions conducive to the site-specific beneficial uses described in Section 2.1.3. Bacteriological conditions along the shoreline are of particular interest for shellfish harvesting and for water-contact recreation, namely swimming and surfing. However, disinfection of MBCSD effluent routinely reduces bacterial densities to levels well below receiving-water standards prior to discharge, and the more-than 100-fold dilution that occurs shortly after discharge renders the plant's bacterial loading imperceptible. Consequently, surfzone bacterial samples are too distant to be materially affected by the discharge, and are instead dominated exclusively by onshore sources and runoff (MRS 1994 – 2010). Therefore, the current NPDES permit no longer requires regular monitoring of surfzone bacteria, but triggers monitoring at eight surfzone locations along Atascadero State Beach only when there is an upset in the WWTP's disinfection process that causes total coliform density in the effluent to exceed 2400 MPN/100 ml.

The surfzone stations are located at gradient distances upcoast and downcoast relative to Station C, which is positioned at the onshore site closest to the offshore discharge location (See Table 3.2 and Figure 3.1). During 2016, as in prior years, no instances of elevated effluent coliform triggered the requirement for surfzone sampling.

Table 3.2 Target Locations of Surfzone Sampling Stations

Station	Description	Along-Shore Distance ¹ (m)	Latitude ²	Longitude
A1	Upcoast Reference	1330 N	35° 23.967' N	120° 52.116' W
A	Upcoast Midfield	912 N	35° 23.750' N	120° 52.067' W
B	Upcoast Nearfield	488 N	35° 23.517' N	120° 52.000' W
C	Onshore of Diffuser	0	35° 23.250' N	120° 51.950' W
D	Downcoast Nearfield	426 S	35° 23.033' N	120° 51.917' W
E	Downcoast Midfield	922 S	35° 22.767' N	120° 51.900' W
F	Downcoast Reference	1602 S	35° 22.400' N	120° 51.883' W
G	Morro Creek ³	— ⁴	—	—

Other beneficial uses designated for this coastal area include fishing and non-water-contact recreation, consisting of beachcombing, boating, and picnicking. These uses were addressed by visual observations of aesthetic conditions, which were documented during the offshore surveys conducted in 2016. Other auxiliary observations included weather conditions, ocean currents, tides, and any evidence of water discoloration, floating oil and grease, turbidity, odor, or materials of sewage origin on the beach or in the water.

¹ Along-shore distance and direction from Station C

² As described in the glossary, all coordinates cited in this report are referenced to the WGS84 datum

³ Immediately before flowing to the ocean

⁴ Location varies with the along-shore migration of the mouth of Morro Creek

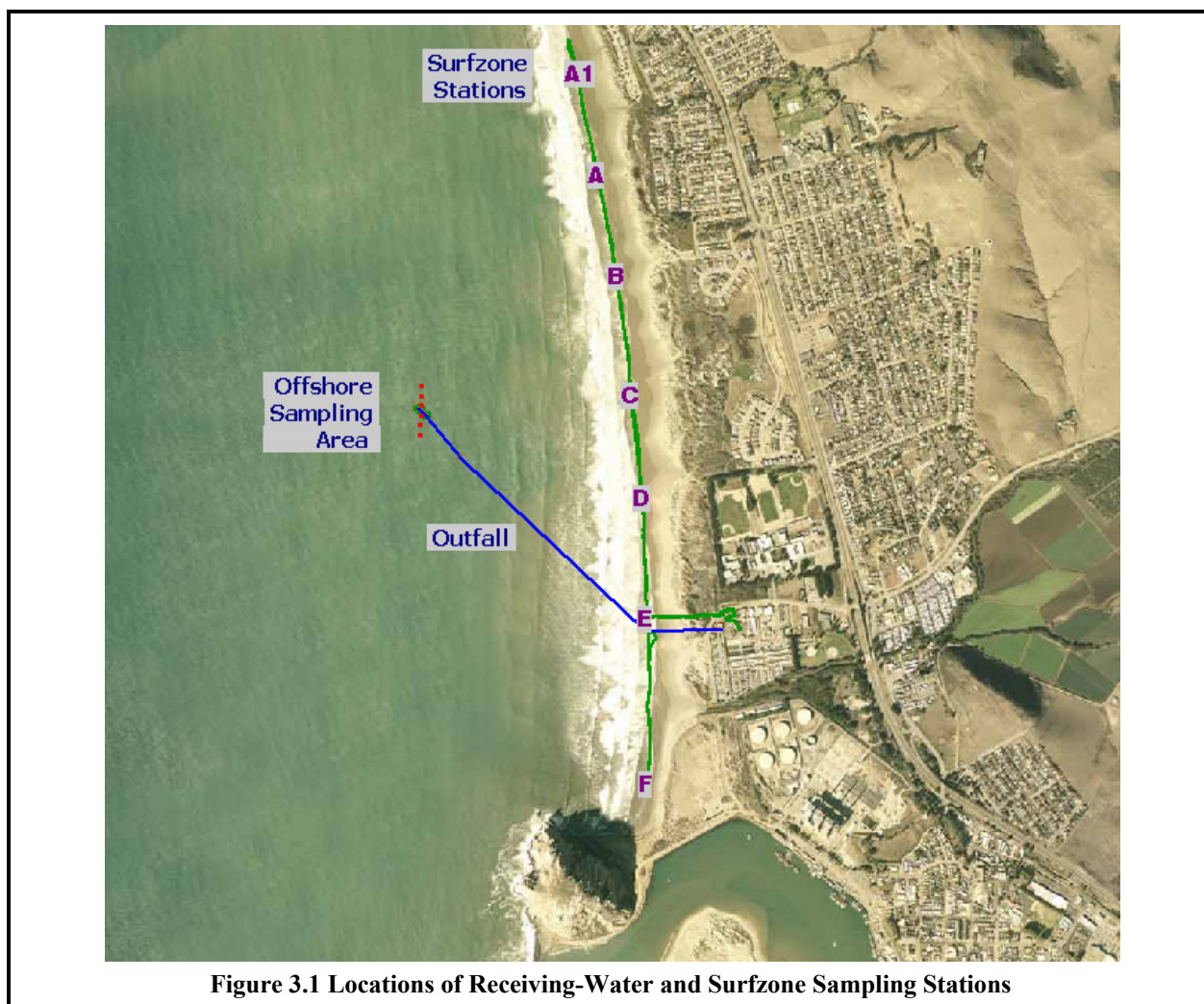


Figure 3.1 Locations of Receiving-Water and Surfzone Sampling Stations

3.1.3 Sampling Station Design

Offshore receiving-water sampling focuses on the water column close to the discharge location (Figure 3.1). The beginning of the diffuser structure lies 827 m from the shoreline, while the diffuser structure itself extends an additional 52 m toward the northwest along the seafloor.

Twenty-eight of the 34 available ports discharge effluent along a 42 m section of the diffuser structure. The diffuser ports were hydraulically designed to dilute effluent rapidly within the receiving seawater immediately upon discharge. Most of this turbulent mixing occurs within a zone of initial dilution (ZID), which by regulatory assertion alone, extends laterally to a distance of approximately 15 m from the center of the diffuser structure.

Over longer periods, energetic waves, tides, and coastal currents within Estero Bay further disperse the effluent plume within open-ocean receiving waters beyond the ZID. Areas of special concern, such as sanctuaries and estuaries, are too distant to be materially affected by the treated wastewater discharge. For example, the southern boundary of the Monterey Bay National Marine Sanctuary is located 38 km to the north, while the entrance to Morro Bay lies 2,800 m to the south of the discharge. In 1995, Morro Bay

was included in the National Estuary Program; however, because of its distance from the discharge, incursion of unmixed wastewater into the Bay is implausible. Direct seawater exchange between the discharge point and Morro Bay is restricted by Morro Rock, which lies between the outfall and the south-facing entrance to the Bay. Morro Rock is the largest physiographic feature along the adjacent coastline and extends into Estero Bay approximately 2,000 m south of the point of discharge (Figure 3.1).

During 2016, vertical seawater measurements were recorded at six offshore stations situated at three gradient distances upcoast and downcoast from the ZID (Figure 3.2). The sampling pattern was designed to evaluate the discharge's region of influence during each quarterly water-quality survey. The two closest stations, which are on the outer margin of the ZID, were located 15 m from the closest point on the diffuser structure. The remaining more-distant stations were located relative to the center point of the diffuser structure.

It is important to consider the “closest-approach” distance when evaluating impacts at stations near the wastewater discharge. Although the discharge has historically been considered a “point source” for most large-scale modeling purposes, it does not occur at a single location of infinitesimal size. Because the discharge is distributed along a 42 m section of the seafloor, its finite size affects assessments of wastewater dispersion at sites close to the discharge. Specifically, the amount of wastewater dispersion at a given point in the water column is determined more by its distance to the closest diffuser port rather than its distance to the center point of the diffuser structure. Near the source, this “closest-approach” distance can be much less than the actual distance to the center of the diffuser structure (Table 3.3, Figure 3.2). As such, it would be inappropriate to design a nearfield sampling pattern based solely on center-point distances rather than closest-approach distances.

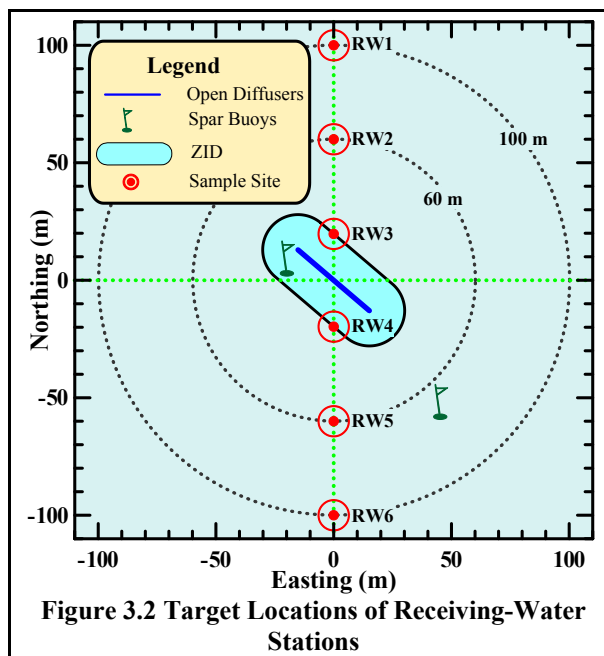


Figure 3.2 Target Locations of Receiving-Water Stations

Table 3.3 Target Locations of Receiving-Water Stations

Station	Description	Latitude	Longitude	Closest Approach Distance ¹ (m)	Center Distance ² (m)
RW1	Upcoast Midfield	35° 23.253' N	120° 52.504' W	88.4	100
RW2	Upcoast Nearfield	35° 23.231' N	120° 52.504' W	49.4	60
RW3	Upcoast ZID	35° 23.210' N	120° 52.504' W	15.0	20
RW4	Downcoast ZID	35° 23.188' N	120° 52.504' W	15.0	20
RW5	Downcoast Nearfield	35° 23.167' N	120° 52.504' W	49.4	60
RW6	Downcoast Midfield	35° 23.145' N	120° 52.504' W	88.4	100

With the closest-approach distances in mind, the six receiving-water monitoring stations were aligned along a north-south axis at the same isobath (15.2 m) as the diffuser center point. Stations RW3 and RW4 were positioned at the boundary of the ZID at a distance of 15 m upcoast and downcoast of the closest

¹ Distance from the closest open diffuser port

² Distance to the center of the open diffuser section

diffuser ports (Table 3.3). Stations RW2 and RW5 were located at nearfield distances (60 m) from the diffuser center point, which positioned them 49.4 m from the closest diffuser port. Midfield Stations RW1 and RW6 represent reference stations situated 100 m upcoast and downcoast of the center point. Depending on the direction of the local oceanic currents at the time of sampling, one or more of these near and midfield stations could conceivably be influenced by wastewater discharge. Under those circumstances, the midfield station on the opposite side of the diffuser acts as a reference station. Comparisons of water properties at these diametrical stations quantify discharge-induced departures from ambient seawater properties that are used in the evaluation of compliance with the NPDES discharge permit.

3.1.4 Navigation

Vessel positioning within the compact offshore sampling pattern improved markedly with the advent of Differential Global Positioning Systems (DGPS). DGPS, with typical position errors of less than 2 m, significantly enhances the accuracy of traditional GPS navigation systems, which have position errors as large as ± 15 m, a span equaling the width of the ZID itself. At the beginning of 1998, the survey vessel F/V *Bonnie Marietta* was fitted with a Furuno™ GPS 30 and FBX2 differential beacon receiver. This navigational system was used on 29 July 1998 to accurately locate the position of the open section of the diffuser structure (MRS 1998b), and to establish the new target locations for the offshore monitoring stations listed in Table 3.3.

Specifying the precise location of each measurement relative to the diffuser is crucial for accurate interpretation of the water property fields. For example, during vertical profiling, the actual sampling locations do not coincide with the exact target coordinates listed in Table 3.3. The combined influences of winds, waves, and currents induce vessel offsets of tens of meters within the minute or so it takes to complete the water-property profile at each station. Even during quiescent seastates, drift is induced by residual vessel momentum remaining after station approach. Prior to the use of DGPS, these offsets could not be reliably resolved by the available navigation, but, during the 2016 surveys, the sampling locations were precisely determined throughout the vertical profiling conducted at each station.

Determination of vessel drift during vertical profiling is an important consideration because it caused some of the measurements in 2016 to be collected within the ZID, where the water-quality objectives in the COP do not apply (see shaded entries in Table 3.4). The average closest-approach distances in Table 3.4 can be compared with the target distances listed in Table 3.3 to assess the fidelity of the sampling locations with respect to diffuser-structure proximity. From the standpoint of an ideal compliance evaluation, all measurements would be made at or slightly beyond the ZID boundary, namely, at distances

Table 3.4 Station Proximity to the Discharge

Station	Description	Closest-Approach Distance ¹ (m)				
		Target	March	May	September	October
RW3	Upcoast ZID	15.0	7.6²	17.9	15.4	11.8
RW4	Downcoast ZID	15.0	23.8	15.3	11.0	8.3
RW2	Upcoast Nearfield	49.4	39.0	42.5	50.3	54.7
RW5	Downcoast Nearfield	49.4	60.3	43.1	50.8	48.0
RW1	Upcoast Midfield	88.4	83.4	94.4	87.1	89.9
RW6	Downcoast Midfield	88.4	97.5	86.9	88.7	91.6

¹ Distance from the closest open diffuser port to the average station position

² All or portions of the profiles highlighted in bold typeface with shaded cells were sampled within the ZID

of more than 15 m from the diffuser structure. However, measurements collected well within the ZID, and close to the diffuser structure, are also valuable because they capture the dynamics of the plume while it is still undergoing rapid initial mixing, and thus lend timely insight into the dispersive capabilities of the outfall. These close-in measurements have demonstrated that the MBCSD discharge routinely meets the COP receiving-water objectives within the ZID and well before completion of the initial dilution process.

3.1.5 Sampling Equipment and Methodology

The 38-ft F/V *Bonnie Marietta*, owned and operated by Captain Mark Tognazzini of Morro Bay, provided vessel support for the four offshore surveys conducted in 2016. Douglas Coats of Marine Research Specialists (MRS) supervised scientific operations as Chief Scientist, and provided data-acquisition and navigational support during all the surveys. He also assisted with the deployment and recovery of the CTD and drifter, and collected meteorological measurements at each station. Crewmember William Skok managed deck operations and collected Secchi Depth measurements at each station. During the September survey, Mr. Bruce Keogh, the MBCSD Wastewater Division Manager, and John Gunderlock, a Grade V operator at the WWTP, monitored operations onboard the vessel as client inspectors; and they also participated in data gathering by collecting Secchi Depth measurements at each station.

Standard observations for weather, seas, water clarity/coloration, Secchi depth, and the presence of any odors and floating debris were recorded during each of the surveys. Wind speeds and air temperatures were measured with a Kestrel® 2000 Thermo-Anemometer during the March and May surveys, and with a handheld Holdpeak 866B Digital Thermo-Anemometer during the September and October surveys.

Auxiliary Measurements

After collection of vertical CTD profiles, each station was re-occupied and a Secchi disk was lowered through the water column to determine its depth of disappearance. This provided a visual measure of the water clarity within the upper water column. The depth of disappearance is inversely proportional to the average amount of organic and inorganic suspended material along a line of sight immediately below the sea surface. As such, Secchi depth is a measure of ambient light penetration, which can be limited during upwelling when plankton density increases within the near-surface euphotic zone. The depth of the euphotic zone, where most oceanic photosynthesis occurs, extends to approximately twice the Secchi depth. Because Secchi depths are less precise than measurements collected with electronic sensors, these and the other qualitative sea-surface observations described above, were ancillary to the digital measurements of seawater parameters collected by the CTD throughout the water column.

At the beginning of each survey, a drifter was deployed near the open section of the diffuser structure. The drifter was drogued at mid-depth (7 m) using the curtain shade design of Davis et al. (1982). In this configuration, the drifter trajectory was largely determined by the oceanic flow field rather than surface winds. Satellite navigation onboard the drifter allowed continuous recording of the precise position of the drifter throughout its deployment. Additionally, the vessel's locations during deployment and recovery of the drifter were recorded for navigational comparison. These trajectory measurements established the ambient flow velocity throughout each survey, and indicated the path of plume transport to aid in the interpretation of seawater property distributions.

Instrumental Measurements

A Sea Bird Electronics SBE-19plusV2 CTD instrument package was used during all four of the 2016 surveys. It was deployed in both a vertical water-profiling mode, as well as a horizontal-tow configuration. It collected measurements of conductivity, temperature, light transmittance, dissolved oxygen (DO), pH, and pressure at a sampling rate of 4 Hz (0.25-second intervals). Submersible pumps on

the CTD continuously flushed water through the conductivity cell and oxygen plenum at a constant rate, independent of the CTD's motion through the water.

The six seawater properties used to assess receiving-water quality in this report were derived from the continuously recorded output of the CTD's probes and sensors. Pressure housing limitations on the oxygen and pH sensors confine the CTD to depths less than 200 m (Table 3.5), which is well beyond the maximum depth of the deepest station in the outfall survey. The precision and accuracy of the various probes, as reported in manufacturer's specifications are also listed in the Table. Salinity (‰, parts-per-thousand) was calculated from conductivity measurements reported in units of Siemens/m. Density was derived from contemporaneous temperature (°C) and salinity data, and was expressed as 1000 times the specific gravity minus one, which is a unit of sigma-T (σ_t).

Table 3.5 Specifications for the CTD Instrument

Component	Depth¹	Units	Range	Accuracy	Resolution
Housing (19p-1a; Acetron Plastic)	680	m	0 to 680	—	—
Pump (SBE 5P)	3400	—	—	—	—
Pressure (19p-2h; Strain-Gauge)	680	dBar	0 to 680	±1.7	± 0.10
Conductivity	3400	Siemens/m	0 to 9.0	± 0.0005	± 0.00005
Salinity	3400	‰	0 to 58	± 0.004	± 0.0004
Temperature	3400	° C	-5 to 35	± 0.005	± 0.0001
Transmissivity (WETLabs C-Star) ²	2000	%	0 to 91.3 ³	± 0.3	± 0.03
Oxygen (SBE 43)	200	%Saturation	0 to 120	± 2	—
pH (SBE 18)	200	pH	0 to 14	± 0.1	± 0.006

All three of the physical parameters (salinity, temperature, and density) helped determine the lateral extent of the effluent plume during the tow phase of the surveys. Additionally, during the vertical-profiling phase, they quantified layering, or vertical stratification and stability of the water column, which affects the behavior and dynamics of the effluent as it mixes with seawater within the ZID. Data on the three remaining seawater properties, light transmittance (water clarity), hydrogen-ion concentration (acidity/alkalinity – pH), and dissolved oxygen (DO), further characterized receiving waters and were used to assess compliance with water-quality criteria. Light transmittance was measured as a percentage of the initial intensity of a transmitted beam of light detected at the opposite end of a 0.25-m path. Increased transmittance indicates increased water clarity and decreased turbidity.

The SBE-19plusV2 CTD system was placed into service in mid-2011 and offers many advantages over the other CTD in the MRS equipment inventory, an SBE-19 Seacat. That unit was successfully used for two decades in previous MBCSD surveys. For example, the 4 Hz sampling rate on the new instrument collects data at twice the rate of the older unit, allowing much higher spatial resolution for a given tow, or descent rate. The probes and sensors on the current CTD also have a much faster response time, further enhancing the spatial resolution of seawater properties. Finally, the probes and sensors on the current CTD unit are more stable and exhibit negligible long-term drift. As a result, and in accordance with the manufacturer's recommendations, the current CTD package does not require regular recalibration of the

¹ Maximum depth limit in meters

² 25-cm path length of red (660 nm) light

³ The transmissometer was calibrated to yield 100% transmission in air, and the maximum transmission of 660nm light in pure water is 91.3%.

sensors prior to each field survey. However, these instrumental improvements and operational differences did not materially affect the temporal compatibility among datasets collected before and after the new CTD was commissioned and placed into service.

After the drifter was deployed during the 2016 surveys, the CTD was held below the sea surface for several minutes for equilibration as the survey vessel was positioned for the first mid-depth tow transect. Prior to deployment, the CTD package had been configured for horizontal towing with forward-looking probes. The protective cage around the CTD was fitted with a horizontal stabilizer wing and a depth-suppression weight was added to the towline to achieve near constant-depth tows. During this first deployment, the CTD was continuously towed around and across the ZID at two separate depths in accordance with the receiving-water monitoring requirements of the current NPDES discharge permit. Monitoring of real-time sensor output was used to evaluate the CTD's progress and adjust the tow depth, which was accomplished by small adjustments to vessel speed.

An example of the tow pattern, from the fourth-quarter survey, is shown in Figure 3.3. During each of the four surveys in 2016, eight transects of mid-depth were collected at a speed near 2 m/s over the span of approximately 40 minutes (blue-green lines in Figure 3.3). Subsequently, eight additional passes were made with the CTD at a shallower depth (orange lines). At the observed towing speeds and the 4 Hz sampling rate, at least two CTD measurements were collected for each meter traversed. This complies with the NPDES discharge permit requirement for minimum horizontal resolution of at least one sample per meter during at least five passes around and across the ZID at two separate depths, one within the surface mixed layer and one at mid-depth within the thermocline. Contemporaneous navigation fixes recorded onboard the survey vessel were adjusted for CTD setback and aligned with time stamps on the internally recorded CTD data. The resulting data for the six seawater properties were then processed to produce high-resolution horizontal maps within the upper mixed-layer and sub-thermocline portions of the water column.

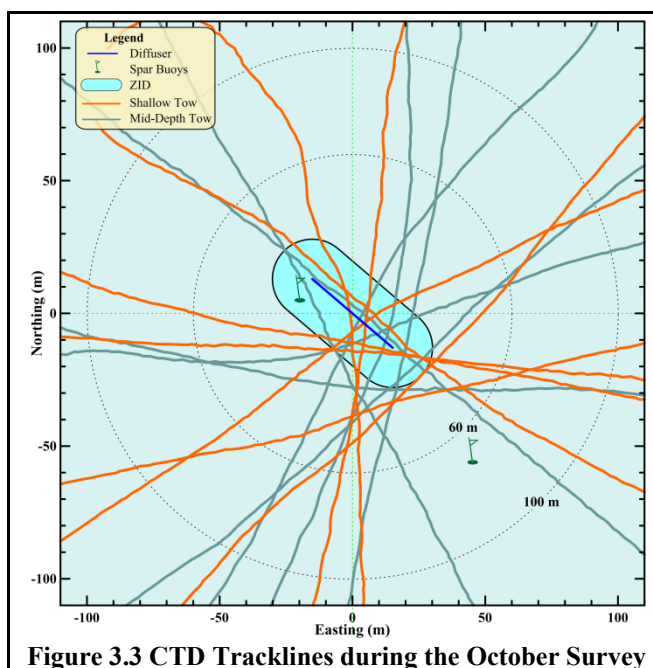


Figure 3.3 CTD Tracklines during the October Survey

Following completion of the last shallow transect, the CTD package was brought aboard the survey vessel and reconfigured for vertical profiling. The CTD was then redeployed and held again beneath the surface to equilibrate while the vessel was repositioned for the first vertical profile. The CTD was then raised to within 0.5 m of the sea surface and profiling commenced. The CTD was lowered at a continuous rate of speed to the seafloor. Measurements at all six stations were collected during a single deployment of the CTD package by towing it below the ocean surface while transiting between adjacent stations. Upon retrieval of the CTD, the data were downloaded to a portable computer and examined for completeness and range acceptability. Data recorded during vertical profiling were processed in accordance with standard analysis procedures (SCBFMC 2002), which resulted in bin-averaged data across 0.5-m standard

depth intervals. The profile data were collated in individual survey reports (MRS 2016cdef) and digitally archived within the USEPA STORET database using their Water Quality Exchange (WQX) framework.

3.2 PLUME DISPERSION

Offshore monitoring is conducted to determine whether the wastewater discharge is causing significant impacts on the receiving waters of Estero Bay beyond the ZID. Section 3.3.1 makes this determination by comparing the water quality at the boundary of the ZID, and gradient areas beyond the ZID, with background seawater properties. However, a complex suite of physical oceanographic processes induces wide variation within the ambient seawater properties near the outfall. This variability makes detection of the relatively minor discharge-related changes challenging. Because of this, the significance of potential discharge-related impacts is evaluated by comparing the amplitude of excursions caused by the presence of dilute wastewater constituents against the inherent variability in ambient seawater properties. Statistically speaking, if the amplitude of the effluent anomaly is small compared to the expected range in ambient seawater properties, then the water-quality impact caused by the discharge cannot be deemed significant.

The following two subsections describe the nature of wastewater dispersion from the MBCSD diffuser structure. The first subsection discusses the configuration of the multiport diffuser structure and describes how its hydraulic design determines the dimensions of the ZID. The second subsection describes the dynamics of the rapid mixing that occurs within the ZID upon discharge from the diffuser structure.

The final two subsections describe oceanographic processes that affect ambient seawater properties within northern Estero Bay and how those properties vary as a result of changes in the oceanographic processes. Quantitative measures of natural-variability are used to establish thresholds for determining whether discharge-induced changes are large enough to impact the beneficial uses of the receiving waters significantly. Regional processes induce major fluctuations in the character of the receiving waters near the diffuser, and large seasonal changes in the vertical structure directly affect the dispersive capability of the outfall.

3.2.1 Zone of Initial Dilution

Treated wastewater from the MBCSD WWTP is discharged into northern Estero Bay through a 1,450-m outfall-diffuser system (Table 3.6). The existing outfall was constructed in 1982 with an up-graded 0.686-m diameter steel pipe lined and coated with cement mortar. The outfall extends to a water depth of about 15.2 m (50 ft), approximately 0.5 nautical miles offshore of Atascadero State Beach. At its terminus is a multiport linear diffuser consisting of 34 ports, each 5.08 cm in diameter. The ports are spaced 1.52 m apart on alternating

Table 3.6 Hydraulic Characteristics of the Outfall

Parameter	Measurement
Outfall diameter (m)	0.686
Outfall length (m)	
Land	207
Ocean	1,449
Diffuser diameter (m)	0.686
Diffuser length (m)	51.8
Port orientation (° from horizontal)	0
Port diameter (m)	0.0508
Orifice contraction coefficient	0.89
Center point depth of outfall ports (m MLLW)	15.2
Number of ports open	28 ¹
Port spacing ² (m)	1.52
Design flow rate for each port (m ³ /s)	0.01 ³

¹ 28 of 34 ports available

² Ports are located on alternating sides of the diffuser. Distance between ports on the same side of the diffuser is 3.04 m.

³ This flow rate applies when 34 ports are open, with a total flow of 0.340 m³/s (7.76 MGD) through the diffuser structure. The outfall design capacity is 0.358 m³/s (8.17 MGD).

sides of the pipe. The current discharge volume from the WWTP only requires the use of 28 of the 34 available ports.

For regulatory and modeling purposes, an idealized ZID is envisioned around the diffuser structure, where the most intense turbulent mixing of effluent is thought to take place shortly after discharge. The ZID is assumed to extend beyond the dimensions of the diffuser structure to a horizontal distance equal to the water depth (USEPA 1994). The water depth at the center point of the diffuser structure is 15.24 m (50 ft) measured relative to the MLLW tidal datum. Consequently, the ZID forms a vertical cylinder whose horizontal cross-section is an elongated ellipse as shown in Figure 3.2 on Page 3-6. Based on the diffuser specifications, the dimensions of the ZID are 73.15 m (240 ft) in length, 31.2 m (102.4 ft) in width, and 15.24 m (50 ft) in height. The length is twice the water depth (15.24 m) added to the 42.67 m (140 ft) long diffuser section where ports are open. The width of the ZID is twice the water depth added to the inside diameter (0.68 m) of the diffuser pipe. Coatings, which add 0.2 m to the outside diameter of the pipe, and the angled diffuser ports, which extend beyond the edge of the pipe, were neglected for the purposes of determining ZID dimensions. Based on these dimensions, the ZID covers an ocean surface area of approximately 3000 m² and contains 12 million gallons of seawater. At the nominal discharge rate of 1 MGD, it would take twelve days for the wastewater discharge to fill a volume equal to the ZID.

In reality, at any given time, the actual initial-dilution zone departs markedly from the idealized vertical cylinder described above. On a regular basis, the prevailing currents carry the effluent plume well beyond the boundaries of the idealized ZID, long before it achieves buoyant equilibrium and well before initial mixing is complete. Nevertheless, because wastewater is rapidly diluted upon discharge, the water-quality objectives of the COP are consistently met, almost immediately upon discharge and well within the boundary of the ZID.

3.2.2 Critical Initial Dilution Ratio

During the first few minutes after discharge, wastewater dilution is determined by the physical characteristics of the diffuser system and stratification within the receiving waters. This initial rapid mixing occurs within the ZID, and close to the diffuser structure. There, the turbulent processes associated with the high-velocity discharge jets emanating from each diffuser port, and the buoyant rise of the effluent plume through the water column, entrain seawater and result in the rapid dilution of discharged wastewater. Subsequent transport and dispersion by regional oceanographic processes further dissipate the effluent, albeit at a slower rate. The dispersion associated with these oceanographic processes is not considered part of initial dilution.

The limiting concentrations for wastewater constituents specified in the NPDES permit (see Chapter 2) were determined by applying the critical initial dilution ratio for the MBCSD outfall to the open-ocean water-quality objectives specified in the COP for individual contaminants (Table B of SWRCB 2005). The critical initial dilution applicable to the MBCSD outfall was conservatively estimated to be 133:1 (Tetra Tech 1992). The following example shows how this dilution factor was applied to determine the wastewater limits that were established for cadmium in the NPDES permit. The COP's water-quality objective for cadmium in receiving waters has an instantaneous maximum concentration of 0.01 mg/L or less (SWRCB 2005). Based on worst-case modeling under extremely stratified conditions, there will be at least 133 parts of ambient water mixed with each part of wastewater after completion of initial mixing. Applying this in the definition of dilution (Fischer et al. 1979), the concentration of a particular contaminant in wastewater is given by:

$$C_e \equiv C_o + D (C_o - C_s) \quad \text{Equation 3.1}$$

where: C_e = the concentration of a constituent in the wastewater effluent,
 C_o = the concentration of the constituent in the ocean after dilution by D ,
 D = the dilution ratio of the volume of seawater mixed with wastewater, and
 C_s = the background concentration of the constituent in ambient seawater.

The background concentration (C_s) for cadmium in seawater is negligible, so the wastewater limit is ($D+1$) times the COP objective (C_o). Thus for cadmium, the instantaneous maximum concentration within wastewater is restricted to 1.34 mg/L, or 134 times the seawater objective of 0.01 mg/L. Specifically, under the worst-case scenario, wastewater containing 1.34 mg/L of cadmium will dilute to a concentration of 0.01 mg/L upon reaching the boundary of the ZID, and the COP receiving-water objective will be met. This demonstrates how the maximum effluent (end-of-pip) limit on cadmium concentrations (1.34 mg/L) was established in the NPDES discharge permit granted to the MBCSD.

The foregoing calculation also reveals how strongly the assumed initial dilution influences the wastewater limitations that are specified in the NPDES discharge permit. Because an unusually low value was specified for the critical initial dilution (133:1), conservative wastewater limitations were established for the MBCSD discharge. As discussed in Section 3.3.1, however, direct observations recorded close to the diffuser during 2015 and 2016 surveys indicate that dilutions much higher than 133:1 were achieved well within the ZID and below the 6.4-m plume-trapping depth identified in the dilution model. In fact, the original plume modeling that established the 133:1 critical dilution (Tetra Tech 1992) included 34 open diffuser ports instead of the 28 ports that are currently in use, thereby underestimating the ejection velocity and the associated turbulent mixing rate achieved by individual diffuser ports.

Additionally, the model assumed that the receiving waters at the point of discharge were unrealistically stratified. The stringent vertical stratification profile used in the modeling is not typical of the receiving-water conditions surrounding the outfall. Specifically, the critical initial dilution computed by the original modeling was overly conservative because the density profile used to establish the MBCSD critical initial dilution was collected 1 km south of the diffuser structure, where the water column was excessively stratified due to the influence of the thermal discharge from the Morro Bay power plant.¹ At the time the “critical” density profile was measured, the power plant was discharging heated water to the ocean near Station F, next to Morro Rock (see Figure 3.1 on Page 3-5).

Because this southerly monitoring station consistently exhibited anomalous water properties relative to the ambient conditions near the MBCSD outfall, it was discarded when the receiving-water monitoring program was revised in 1999. Nevertheless, the original plume modeling, which established both the critical initial dilution ratio currently applied to the outfall and the present effluent discharge limits, incorporated an artificially stratified profile from this station. The model predicted that the receiving waters would trap the effluent plume 6.4 m below the sea surface, which is unnaturally deep because of the unrealistic density profile used in the model. Accordingly, more recent dilution modeling demonstrates that with weaker vertical stratification representative the water column near the diffuser, a critical initial dilution of approximately 200:1 with a plume trapping depth of 1.4 m best characterizes the worst-case outfall performance (Lindstrom 1998). These recent dilution modeling results are consistent

¹ The plant has not operated for several years and is now permanently closed by its current owner, Dynegy Corporation. However, at the time the original initial dilution modeling was performed, it was discharging a large volume of heated seawater north of Morro Rock.

with the observations collected in field surveys conducted since the inception of the MBCSD offshore monitoring program, including the results described in this report.

Nevertheless, even with dilutions of only 133:1, the highly sensitive instruments on the CTD have a difficult time detecting the direct influence of the dilute wastewater constituents, except very close to diffuser ports. Accordingly, most wastewater-induced perturbations in water quality are imperceptible beyond the ZID, particularly in the presence of the large, natural oceanic variability that exists within northern Estero Bay.

3.2.3 Regional Oceanographic Processes

A broad suite of complex oceanographic processes determines the disposition of coastal marine waters within northern Estero Bay. Close to shore, an energetic wave field occasionally dominates the oceanographic conditions along this section of the central California coast. This wave energy increases the turbulence within nearshore waters, and causes significant sediment resuspension and sand transport along the outfall corridor. Farther offshore, bands of counter currents move ocean waters in alternating directions along the continental shelf.

These large-scale, along-shore current flows are interrupted by cross-shore tidal currents and by turbulent jets that occasionally form at major promontories, such as Point Piedras Blancas, Point Buchon, and Point Arguello (Figure 3.4). These narrow, energetic jets transport materials far from the coast, and are repeatedly observed extending from upwelling centers located south of major coastal headlands in satellite imagery. In addition to these processes, wind-driven upwelling alters the vertical distribution of water properties within Estero Bay and contributes to the remarkably high productivity of the marine waters along the central California coast.

These open-ocean flow processes heavily influence the character of, and variability in sea-surface water properties within Estero Bay. Beyond the continental slope (>50 miles offshore), the diffuse, southward-flowing California Current represents the eastern limb of the clockwise-flowing gyre that covers much of the North Pacific Ocean. Before turning south to form the California Current, subarctic water is carried along high latitudes where it is exposed to atmospheric cooling, nutrient regeneration, and precipitation. As a result, the waters of the California Current are characterized by a seasonably stable low salinity (32‰ to 34‰), low temperature (13°C to 20°C), and high nutrient concentrations. Because of this stabilizing effect, seawater off the central California coast undergoes less seasonal variation than surface waters at similar latitudes along the eastern seaboard.

The northward flowing Davidson countercurrent lies along the continental slope and shelf immediately shoreward of the California Current. It carries more saline and less oxygenated seawater from the Southern California Bight. The Davidson Current exhibits marked seasonal variability in its flow depth

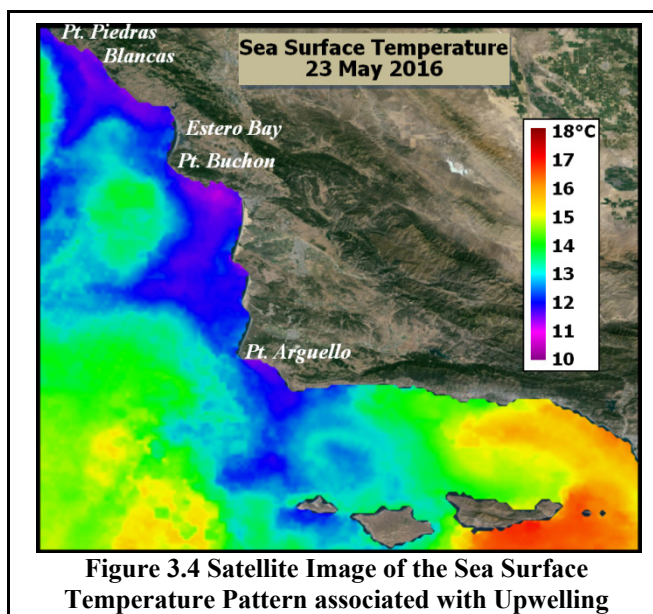
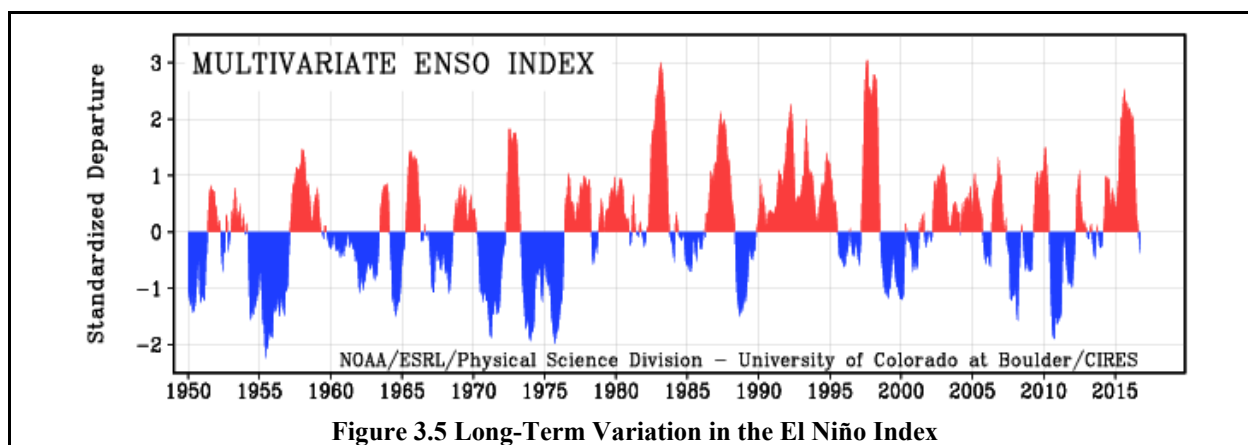


Figure 3.4 Satellite Image of the Sea Surface Temperature Pattern associated with Upwelling

and intensity that is driven by large-scale changes in coastal winds. Specifically, throughout much of the year, winds along the central coast blow from the northwest, parallel to the coast and in opposition to the northward-flowing Davidson Current. These winds cause upwelling along the coastline, which produces the distinctive surface cooling shown in satellite imagery (dark blue and purple shading in Figure 3.4). During these periods, the northward flowing Davidson Current remains at depth, and forms an undercurrent. When the northwesterly winds relax, usually between December and February, the Davidson Current strengthens and can extend to the sea surface.

The spring transition back to strong southwestward winds typically occurs rapidly between March and May, and relegates the Davidson current to the deep offshore waters. Despite reversals observed elsewhere along the California Coast, the Davidson current maintains a sustained subsurface northward flow in the waters offshore Estero Bay (Coats et al. 1991). During the upwelling season, when strong northwesterly winds cause widespread upwelling along the central California coast, warm surface seawater near the coast is transported offshore by the prevailing winds. At these times, upwelling draws the deep Davidson seawater shoreward along the seafloor, and its distinct properties are often observed in the lower water column within in the MBCSD survey area. The Davidson watermass consists of cold, saline, low-DO, and nutrient-rich seawater that is brought northward from the southern California bight with the undercurrent.

On longer time scales, significant inter-annual fluctuations in oceanographic properties and marine fauna have also occurred along the central California coast, and have strongly influenced the waters of Estero Bay. The sequence and magnitude of these long-term events over the last half-century are captured in Figure 3.5. The warm-seawater fluctuation is called the El Niño–Southern Oscillation (ENSO), which cycles at a period of 3 to 5 years (Graham and White 1988). El Niño and its counterpart, La Niña, are oscillations of the ocean-atmosphere system in the tropical Pacific. El Niño is characterized by unusually warm sea surface temperatures and La Niña by unusually cool temperatures in the equatorial Pacific Ocean. The red excursions in Figure 3.5 reflect times when El Niño conditions prevail, while La Niña conditions are shown in blue.



The El Niño event that was occurring in the years before, and at the time of the 2016 surveys (rightmost red region in the Figure) ranks among the top three events of the last half-century. During strong El Niño periods, basin-wide changes in the dynamic balance of wind-driven currents result in modified flow patterns along the coastline of western North and South America (Chelton et al. 1982). Changes in the study region include an anomalous strengthening of Davidson Current outflow from the Southern

California Bight. El Niño events also generally coincide with increased winter storm activity, reductions in zooplankton biomass, and the introduction of marine organisms typically found far to the south.

Conversely, during La Niña events, high pressure builds in the eastern equatorial Pacific while low pressure develops to the west, producing a stronger equatorial pressure gradient. The easterly trade winds strengthen, causing upwelling off the coastlines of North America, Peru, and Ecuador to intensify, and causing a reduction in sea surface temperatures throughout the Eastern Pacific Ocean, including along California's central coast. The low sea surface temperatures along the equator also cause the Pacific storm track to shift northward, typically resulting in drier conditions throughout the southern United States. However, the rainfall pattern does not always correlate with the long-term ENSO index. The exceptionally large rainfall amount experienced within southern California at the beginning of 2017 coincided with the onset of a La Niña event, while the intense El Niño event in the years prior to that time, coincided with a prolonged drought in California.

In addition to, and superimposed on these large-scale oceanic flows are a variety of transient phenomena including intense eddies, swirls, filaments, meanders, and narrow jets of flow. Some of these cross-shore turbulent features are evident in the complex surface-flow pattern depicted in satellite images. For example, in Figure 3.4, upwelling-induced reductions in coastal sea-surface temperature, delineated in shades of dark blue and purple, stretch southward along the coast. South of Pt. Piedras Blancas, Pt. Buchon, and Pt. Arguello, offshore-directed jets carried the cool water offshore. These types of cross-shore flow features are capable of transporting significant quantities of coastal heat, nutrients, and pollutants to offshore waters (Savoie et al. 1991). As these features extend offshore, they are entrained within eddies and swirls that bend the jets in various directions.

Tidal currents provide another mixing mechanism for ocean waters in the study region, although they are not responsible for significant net transport. Estero Bay experiences astronomical tides of diurnal inequality, meaning that the two daily sets of tidal extrema are of unequal amplitude. Tidal fluctuations of as much as 2.1 m can combine with storm surges of up to 0.3 m and result in water-level changes that are a significant fraction of the 15.2 m MLLW depth at the outfall location. Typically, however, the tidal range near the outfall site is approximately 1 m between mean high and low water. Even so, variation in tidal levels among surveys account for noticeable differences in the number of 0.5-m depth bins reported for the standardized vertical datasets.

At shorter periods, shoaling internal waves and surface gravity waves also serve to mix coastal water properties in both the horizontal and vertical directions within the receiving waters. This region of the central coast is relatively unprotected and experiences a comparatively high flux of wave energy, especially in winter when distant North Pacific storms generate episodic swell events that can travel great distances. Under certain conditions, this wave energy impinges on Atascadero State Beach at oblique angles, generating substantial alongshore transport within the coastal littoral cell between Point Estero and Morro Rock (Figure 3.6). When shoaling waves are large, this wave-generated littoral cell can extend well offshore into Estero Bay, and encompass the MBCSD discharge location.

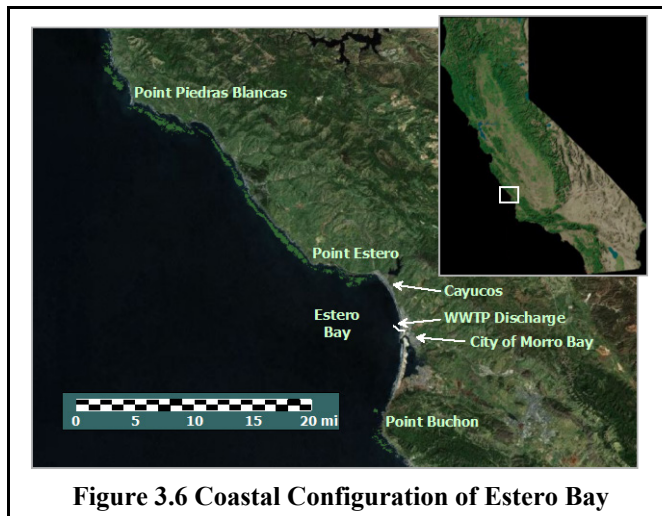


Figure 3.6 Coastal Configuration of Estero Bay

Four primary meteorological sources generate waves along the south-central California coastline: 1) extratropical winter cyclones in the northern hemisphere, 2) northwesterly winds after the spring transition to upwelling conditions, 3) tropical disturbances off shore of Mexico, and 4) extratropical storm swells generated in the southern hemisphere during summer. The first two are the dominant influences on the wave climate within northern Estero Bay, while the last two generate swells from the south, which are diminished near the outfall site by the sheltering effects of Point Buchon (Figure 3.6).

Winter storm waves generated by extratropical winter cyclones are often accompanied by locally severe weather along the south-central coast. These extratropical storms are associated with low-pressure systems that develop along the polar front in the North Pacific Ocean, and that propagate eastward toward the central coast. Thus, major wave events often coincide with an increased discharge from the outfall due to heavy rainfall and increased I&I. These storms occur predominantly from December through March. In fact, more than 95% of all major deepwater wave events ($H_s > 4$ m) occur between the months of November and April (Everts Coastal 1996).

With the exception of major storm events, prevailing northwesterly winds are the predominant mechanism for generating waves near the outfall. These winds dominate during the spring and summer when a high-pressure system is established over the eastern North Pacific Ocean. The winds are highly coherent over this section of the central coast and, consequently, can generate wind waves over a large fetch (Chelton et al. 1987). However, these locally generated waves tend to be of shorter period and smaller height than those generated by major winter storms.

3.2.4 Ambient Seawater Properties

A detailed understanding of ambient seawater properties, and their natural variation within the region surrounding the outfall, is an integral part of compliance evaluation. The receiving-water limitations in the NPDES permit echo the objectives listed in the COP (Table 3.1 on Page 3-3), and the Central Coast Basin Plan (RWQCB 1994). The offshore receiving-water surveys are designed to assess compliance with objectives dealing with undesirable alterations to six physical and chemical characteristics of seawater. Other components of the monitoring program, described in other chapters of this report, address the remaining permit limits.

For the receiving waters, the NPDES permit states that wastewater constituents within the discharge shall not cause the limits listed in Table 3.7 to be exceeded (See Sections V.E through V.K of the permit). The first two receiving-water limits, P1 and P2, rely on qualitative visual observations for compliance evaluation. Compliance with the remaining four receiving-water limitations can be quantitatively evaluated by comparing instrumental measurements with the specific numerical limits listed in the NPDES permit. For example, the numeric limits on absolute values of DO (> 5 mg/L) and pH (7.0 to 8.3), which reflect Basin Plan objectives, can be directly compared with field measurements within the dilute wastewater plume.

However, both P5 and P6 also contain narrative statements, which originate in the COP and define unacceptable water-quality impacts as “*significant*” excursions beyond those that occur “*naturally*.” Quantitative evaluation of these limits requires a more sophisticated comparison between field measurements and numerical thresholds that reflect the natural variation in transmissivity, DO, and pH within the receiving waters surrounding the outfall. The 30-year record of quarterly receiving-water surveys conducted as part of the MBCSD monitoring program provides the necessary insight to rigorously establish compliance thresholds based on natural temporal and spatial variation in ambient seawater properties close to the outfall.

Table 3.7 Permit Provisions Addressed by the Offshore Receiving-Water Surveys

Limit#	Provision	Source	Limit
P1	V.E	COP C.1	Floating particles or oil and grease to be visible on the ocean surface
P2	V.F	COP C.2	Aesthetically undesirable discoloration of the ocean surface
P3	V.I	Basin Plan	Temperature of the receiving water to adversely affect beneficial uses
P4	V.G	COP C.3	Significant reduction in the transmittance of natural light at any point outside the initial dilution zone
P5	V.J	Basin Plan COP D.1	The DO concentration outside the zone of initial dilution to fall below 5.0 mg/L or to be depressed more than 10 percent from that which occurs naturally
P6	V.K	Basin Plan COP D.2	The pH outside the zone of initial dilution to be depressed below 7.0, raised above 8.3, or changed more than 0.2 units from that which occurs naturally

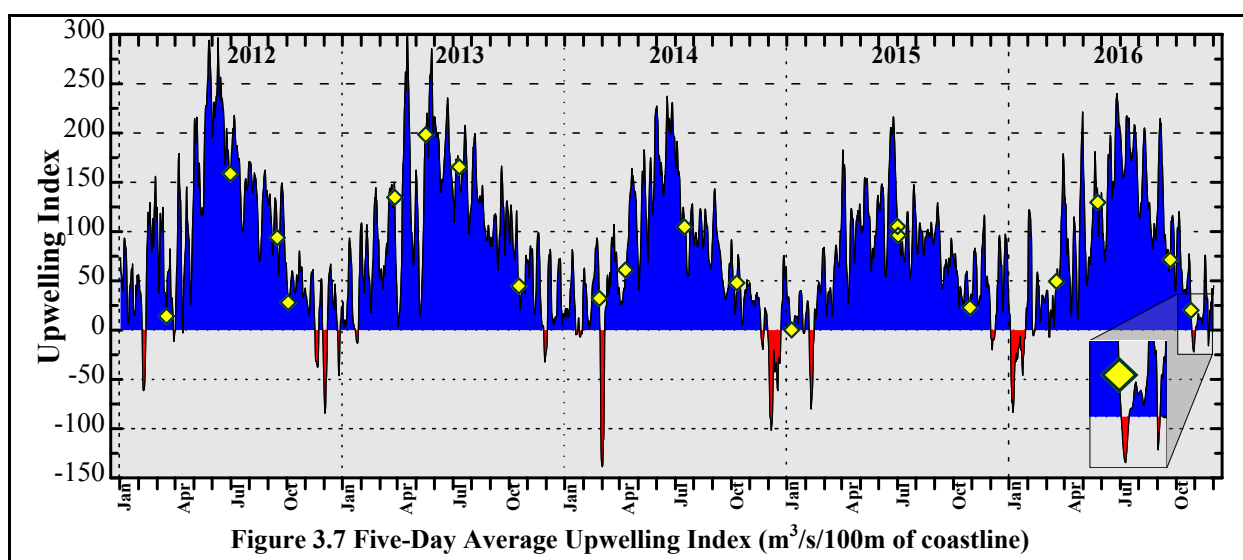
Natural variation in seawater properties is driven by a varying combination of the oceanographic processes described in the previous section. These processes determine the range in ambient seawater properties caused by spatial variation within the survey region at a given time (e.g., vertical stratification), and by temporal variations caused by seasonal and interannual influences (e.g. El Niño and La Niña). Of particular interest are upwelling and downwelling processes that not only determine average properties at a given time, but also the degree of water-column stratification, or spatial variability, present during any given survey. An accurate characterization of stratification helps distinguish discharge-related changes that arise from the presence of wastewater constituents, and thus are subject to the compliance evaluation, from changes that arise because of the upward transport of ambient seawater, which are specifically excluded from the compliance evaluation.

Most of the variability in the ambient seawater database arises from temporal changes in average seawater properties among surveys. Within individual surveys, vertical differences in seawater properties far exceed lateral differences, except during winter surveys when downwelling events can create a uniform water column. The dominating influence of seasonal variability on receiving-water properties, in terms of differences in both vertical stratification and average properties, is reflected in the conditions observed during the four offshore surveys conducted in 2016 (Table 3.8). The Table shows that the upwelling intensity that prevailed at the time of each survey tended to influence the strength of the stratification.

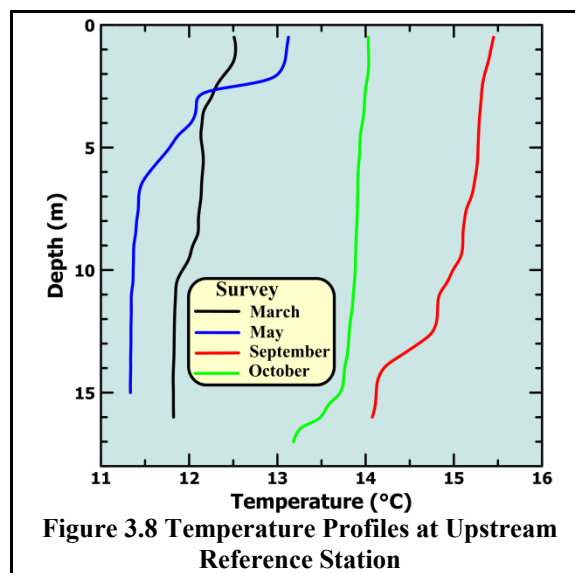
Table 3.8 Oceanographic Conditions during the Quarterly Receiving-Water Surveys

Conditions	March (winter)	May (spring)	September (summer)	October (fall)
Tide	Ebb	Flood	Flood	Ebb
Wave Height (from)	2-3 NW	2-3 NW	1-2 NW	2-3 NW
Flow Direction (toward)	SW	SE	SE	NE
Flow Speed (cm/s)	4.1	3.6	4.2	3.4
ZID Residence Time (min)	6.0	7.0	6.0	7.5
Stratification	Limited	Strong	Moderate	Limited
Water Clarity	Low	Moderate	High	High
Upwelling	Weak	Intense	Moderate	Weak
Plume Visible	Yes	No	No	Yes
Plume Signature	Yes	Yes	Yes	Yes

All four of the 2016 surveys were conducted when some degree of upwelling and water-column stratification was present. Blue shaded areas in Figure 3.7 represent times when northwesterly winds prevailed near Estero Bay, driving surface waters offshore, and inducing upwelling near the coast. The yellow diamonds identify the upwelling intensity at the specific time of the individual receiving water surveys. As seen in the figure, receiving-water surveys over the last several years have generally been conducted during periods of moderate upwelling. Occasionally, however, the first quarter (winter) surveys have been conducted when little or no upwelling was occurring, or when downwelling from successive winter storms produced a nearly uniform water column. During 2016, both the March and October surveys were conducted when upwelling winds were weak, and the October survey was followed by a downwelling event (inset in Figure 3.7). Downwelling displaces deep watermasses offshore, where they no longer provide a nearshore vertical contrast with surface seawater properties, and a uniform (unstratified) water column is produced within the survey area.



The degree of is evident in temperature profiles collected during each of the quarterly surveys in 2016 (Figure 3.8). The profiles were measured at locations upstream of the diffuser structure and thus were not affected by the discharge plume. During the March and October surveys (black and green lines), when upwelling was minimal, seawater temperature was nearly uniform throughout most of the water column. In contrast, temperature differences exceeding 1.5°C were present within the water column during the May and September profiles (blue and red). Over half to the 2°C temperature change during the May survey occurred across only 1 m of depth change. This very sharp thermocline separated the warmer seawater within the surface mixed layer above 2.5 m from a deep cold watermass below 7 m, where the nearly uniform seawater temperature was the coldest measured during 2016, colder even than the



temperatures measured during the winter (March) survey shown in black. This cold watermass was transported shoreward from deep offshore by intense upwelling that prevailed at the time of the survey. The resulting temperature profile is typical of intense upwelling conditions when northwesterly winds along the central California coast drive surface waters offshore, allowing cooler, denser, saltier, and oxygen-poor water at depth to migrate shoreward along the seafloor to replace it.

Although upwelling is recognized as the primary driver behind the productive fisheries of the central California coast, its persistence can also result in depleted oxygen levels within the bottom waters of Estero Bay. Within the euphotic zone in the upper water column, upwelling conditions facilitate phytoplankton blooms that produce oxygen and consume carbon dioxide. However, as phytoplankton die and fall to the seafloor, their decomposition consumes oxygen within deep offshore water masses. Over a period of time, DO levels within the submerged seawater can become depleted due to the absence of recent atmospheric equilibration. When upwelling processes transport this deep watermass shoreward and into the survey area, markedly reduced DO concentrations are occasionally observed near the seafloor. During the 2016 water-quality surveys, DO levels measured in ambient seawater at depth dropped to 5.6 mg/L, and approached the Basin Plan limit of 5 mg/L. In prior years, DO concentrations within ambient seawater at depth fell well below the Basin Plan limit (MRS 2011).

3.3 COMPLIANCE EVALUATION

This section evaluates the significance of potential receiving-water impacts from MBCSD effluent discharged in 2016 through a comparison with the six permit limits listed in Table 3.7. The water-quality limitations are based on criteria in the COP (Table 3.1), the Central Coast Basin Plan, and other state and federal policies that were designed to protect marine life and beneficial uses of ocean waters. Because the limits only pertain to changes in water properties that are caused by the presence of wastewater constituents beyond the ZID, instrumental measurements undergo a series of screening procedures prior to numeric comparison with the NPDES permit thresholds.

The quantitative analyses described in this section focus on water-property excursions caused by the presence of wastewater constituents beyond the ZID, and whose amplitudes can be reliably discerned against the backdrop of ambient fluctuations in seawater properties. A detailed understanding of ambient seawater properties, and their natural variability within the region surrounding the outfall, is therefore, an integral part of the compliance evaluation presented in this section.

These analyses of data collected during 2016 demonstrate that the MBCSD discharge complied with the NPDES discharge permit, as has been the case for the prior quarter century of high-resolution monitoring. Moreover, although observations within the ZID are not subject to compliance evaluations, they often meet the prescribed limits because dilution levels routinely exceed the conservative design specifications assumed in the NPDES discharge permit. Thus, the quantitative evaluation described in this section documents an outfall and treatment process that was continuing to perform at a high level during 2016.

3.3.1 Screening for Applicability

Evaluating whether any of the 46,688 receiving-water measurements collected during 2016 exceeded a permit limit can be a complex process. For example, apparently significant excursions in an individual seawater property are usually unrelated to the presence of wastewater constituents, and instead are due to any number of confounding factors. These confounding factors include natural processes, statistical uncertainty, instrumental error, entrainment of ambient bottom waters within the rising effluent plume,

ongoing initial mixing within and beyond the ZID, and other anthropogenic influences unrelated to the discharge, such as dredging discharges or oil spills.

Because of this complexity, prior to comparison with Basin-Plan numerical limits and COP objectives, measurements are sequentially screened to restrict attention to: 1) the oceanic area where permit provisions apply; 2) changes due to the presence of wastewater particulates; and 3) changes large enough to be reliably detected against the backdrop of natural variation (Table 3.9).

Table 3.9 Receiving-Water Measurements screened for Compliance Evaluation

Topic Addressed	Screening Question	Answer		Parameter
		No	Yes ¹	
Location	1. Was the measurement collected beyond the 15.2-m ZID boundary where modeling assumes that initial dilution is complete?	5,722	40,966	All
Wastewater Constituents	2. Did the beyond-ZID measurement coincide with a quantifiable salinity anomaly ($\leq 550:1$ dilution level) indicating the presence of detectable wastewater constituents?	40,605	361	All
Natural Variation	3. Did seawater properties associated with a quantifiable salinity anomaly depart significantly from the expected range in ambient seawater properties present at the time of the survey?	361	0	Temperature
		339	22	Transmissivity
		361	0	DO
		361	0	pH

Measurement Location

The COP states that compliance with its receiving-water objectives “*shall be determined from samples collected at stations representative of the area within the waste field where initial dilution is completed.*” Initial dilution includes the mixing that occurs from the turbulence associated with both the ejection jet, and the buoyant plume’s subsequent rise through the water column. Although currents often transport the plume beyond the ZID before the initial dilution process is complete, the COP states that dilution estimates shall be based on “*the assumption that no currents, of sufficient strength to influence the initial dilution process, flow across the discharge structure.*” Because of this, the regulatory mixing distance, which is equal to the 15.2-m water depth of the discharge, provides a conservative boundary to screen receiving-water data for subsequent compliance evaluation. Application of this initial screening question to the 2016 dataset eliminated 5,722 of the original receiving-water observations from further consideration because they were recorded within the ZID (Table 3.9, Question 1).

Presence of Wastewater Constituents

The NPDES discharge permit also restricts application of the numerical receiving-water limits to excursions caused by the presence of wastewater constituents. This confines the compliance analysis to changes caused “*as the result of the discharge of waste,*” as specified in the COP, rather than to anomalies that arise from the movement of ambient seawater that becomes entrained in the rising effluent plume. Analyses of high-quality data collected during quarterly receiving-water surveys over the past decade have demonstrated that the direct influence of dilute wastewater is almost never observed in any seawater property other than salinity, except very close (<1 m) to a diffuser port and within its ejection jet.

¹ Number of remaining CTD observations of potential compliance interest based on this screening question

In fact, negative salinity anomalies are the only consistent indicator of the presence of wastewater constituents within receiving water. Wastewater salinity is negligible compared to that of the receiving seawater, so the presence of a distinct salinity minimum provides *de facto* evidence of the presence of wastewater constituents. Because of the large contrast between the nearly fresh wastewater and the salty receiving water, low salinity provides a powerful tracer of dilute wastewater that is unrivaled by other seawater properties. The contrast between effluent and seawater temperature, DO, pH, and transmissivity is much smaller, and as such, their wastewater signatures dissipate rapidly upon discharge with very little mixing. Wastewater's near lack of salinity, however, allows the presence of effluent constituents to be identified even after being diluted to levels substantially greater than the 133-fold critical initial dilution assumed in the NPDES discharge permit.

The association between negative salinity anomalies and wastewater constituents is so robust that the dilution achieved at a given point within the effluent plume can be determined directly from the amplitude of the associated salinity reduction. By rearranging Equation 3.1, the dilution ratio (D) can be computed from the salinity anomaly ($A = C_o - C_s$) as follows:

$$D \equiv \frac{(C_e - C_o)}{(C_o - C_s)} \approx \frac{-C_o}{(C_o - C_s)} \propto A^{-1} \quad \text{Equation 3.2}$$

where:

D = the dilution ratio of the volume of seawater mixed with wastewater,
 C_o = the salinity of the wastewater-seawater mixture after dilution by D ,
 C_s = the background seawater salinity, and
 $A = C_o - C_s$ = the salinity anomaly.

The salinity concentration measured within MBCSD effluent (C_e) is small compared to that of both the receiving seawater (C_s) and, after dilution by more than 100-fold, the salinity of the effluent-seawater mixture (C_o). After dilution, C_o is nearly constant and close to that of ambient salinity (C_s). Consequently, its variation can be neglected in the equation and, to a close approximation; dilution levels are directly proportional to the inverse of the salinity anomaly itself (A). Thus, reduced effluent dilution at a given location within receiving waters is directly reflected by a larger-amplitude salinity anomaly.

The smallest reduction in salinity that can be reliably detected within receiving waters is 0.062‰, which reflects a dilution level of 542-fold when substituted into Equation 3.2. This plume-detection threshold was determined from a statistical analysis of the naturally occurring variability in salinity readings measured near the outfall over a five-year period between 2004 and 2008. Thus, wastewater that has mixed with 542 parts of seawater no longer has a salinity signature that can be reliably discerned against the backdrop of natural variation. This dilution threshold was rounded up to 550-fold for added conservatism in the screening analysis (Table 3.9, Question 2).

The 550:1 plume-detection threshold is more than four-times higher than the 133:1 minimum initial dilution ascribed to the worst-case outfall performance, and upon which compliance with the Table B receiving-water objectives of the COP is ensured through specification of end-of-pipe limits on effluent contaminant concentrations in the NPDES discharge permit. Consequently, discernible salinity anomalies are more than capable of detecting out-of-compliance measurements within receiving waters, while measurements with anomaly amplitudes below the plume-detection threshold have wastewater concentrations far too dilute to be of compliance interest.

Application of this screening question to the 40,966 observations that were measured outside the ZID during 2016 surveys determined that only 361 measurements were associated with reductions in salinity that could potentially be related to the presence of dilute wastewater constituents (Table 3.9, Question 2).

Natural Variation

Another integral part of the compliance evaluation is determining whether a particular measurement differed from other observations at the same depth level because of naturally occurring variations in seawater properties at the time of the survey, or whether it was perceptibly altered by the presence of wastewater constituents (Table 3.9, Question 3). Thus, natural variability must be accurately quantified for each survey as part of the compliance evaluation.

The natural range in ambient seawater properties differs among surveys, largely because of differences in upwelling-induced vertical stratification. It is important to consider this vertical variation when evaluating whether a particular measurement extends beyond its natural range. Departures (anomalies) from background conditions are traditionally defined as the difference between a given measurement and the average of seawater properties measured at the same depth level, but at locations beyond the discharge's influence. However, when the water column is highly stratified, ambient seawater properties near the seafloor differ from those within the rest of the water column, and their juxtaposition within the rising effluent plume can appear as a lateral anomaly at mid-depth. These entrainment-generated anomalies regularly appear within the rising effluent plume when the water column is strongly stratified by upwelling. However, these entrainment-generated anomalies are unrelated to both the concentration of wastewater constituents within a given measurement, and the quality of the effluent that is being discharged at that particular time.

As described in Section 3.2.4, only significant departures in temperature, light transmittance, DO, and pH are relevant for evaluating compliance with receiving water limits. Thresholds for evaluating whether any of these measured properties extended beyond the natural conditions

Table 3.10 Thresholds for Significant Departures from Natural Conditions

Water Property	March	May	September	October
Temperature (°C)	>13.34	>13.75	>16.21	>14.82
Transmissivity (%)	<57.1	<68.7	<70.8	<71.2
DO (mg/L)	<4.26	<5.49	<5.34	<5.40
pH (minimum)	<7.802	<7.842	<7.864	<7.823
pH (maximum)	>8.036	>8.193	>8.129	>8.057

were determined for each receiving-water survey conducted during 2016 (Table 3.10). Excursions beyond the thresholds identify measurements that warrant comparison with the specific numeric thresholds identified in permit provisions (Table 3.7).

The thresholds of natural variation for each water property were established using a two-step procedure analogous to the guidance in COP Appendix VI. First, one-sided 95% confidence bounds on transmissivity (-10.2%), temperature (+0.82°C), DO (-1.4 mg/L), and pH (± 0.94) were determined from an analysis of the same five-year database used to characterize within-survey salinity variation. These bounds quantify the spatial variability inherent in the ambient receiving waters surrounding the outfall at any given time. Second, the confidence bounds were combined with 95th percentiles determined from background seawater properties for each individual survey during 2016, thereby capturing seasonal differences among surveys, including the degree of vertical stratification. The percentiles were determined from vertical profiles excluding measurements potentially affected by the discharge. The resulting set of thresholds quantifies when fluctuations could be deemed a significant departure from natural conditions at the time of each survey. Because the thresholds incorporate ambient vertical

variability, many of the entrainment-generated anomalies artificially created by the upward movement of the plume during stratified conditions are excluded from further compliance consideration, and attention is focused only on potentially significant anomalies caused by the presence of wastewater constituents themselves.

Application of these thresholds to the temperature, pH, and DO concentrations associated with the 361 remaining measurements of potential compliance interest were all well within their respective ranges of natural variability (Table 3.9, Question 3). As such, the screening procedure unequivocally eliminated those measurements from further consideration.

Measurements of Potential Compliance Interest

Because of the high performance of the treatment process and diffuser system, the screening procedure described above usually eliminates all the measurements collected in a given survey from further consideration in the compliance analysis. This is often the case, regardless of whether samples were collected within the ZID or were associated with low salinity anomalies. However, in a highly unusual circumstance, 22 low-transmissivity observations from the March 2016 survey¹ were identified for further compliance analysis (Table 3.9, Question 3) because they ranged well below that survey's 57.1% threshold for natural variability (first data column of Table 3.10).

However, further investigation demonstrated that none of these observations constituted an exceedance of Permit Provision *P4* in Table 3.7 because all were measured beneath the euphotic zone (MRS 2016c). Specifically, their increased turbidity could not have reduced "*the transmittance of natural light*" because the penetration of ambient sunlight was attenuated within the upper water column well before reaching the depth where the measurements were recorded. Furthermore, the suspended-solids loads discharged by the WWTP at the time of the survey were too low to cause the turbidity increases within the 22 low-transmissivity measurements and thus, they did not arise "*as a result of the discharge of waste.*"

Instead, the reduced transmissivity found at mid-depth within the dilute effluent plume during the March 2016 survey resulted from a physical process unrelated to the presence of wastewater constituents. The markedly reduced transmissivity found at mid-depth was caused by upward transport of turbid ambient seawater that was entrained within the buoyant plume immediately after effluent was discharged near the seafloor. Seawater near the seafloor is often naturally higher in turbidity, but when strong bottom currents increase turbulence and shear along the seafloor, they resuspend higher volumes of surficial sediments and lightweight flocs of detritus. This resuspended material forms a very thin, transient, particle-rich layer known as a benthic nepheloid layer (BNL). BNLs are a widespread phenomenon on continental shelves (Kuehl et al. 1996) and have been regularly documented in past surveys conducted within Estero Bay.

During 2016, highly turbid BNLs were present during both the March and October surveys, and transmissivity reductions that "significantly" departed from range in natural conditions were identified as a result (MRS 2016cf). In addition to the 22 low-transmissivity measurements discussed above, the October-2016 database contained two unusually low transmissivity observations near 65.3% that ranged well below the calculated natural variability threshold of 71.2% associated with that survey (fourth data column of Table 3.10). Those two observations were measured immediately above the seafloor and within the BNL that was present throughout the region at the time of the survey. Because they were not associated with low salinities that would identify them as part of the dilute effluent plume, they were

¹ The 22 measurements had transmissivities ranging from 45% to 51%.and were collected along two transects at depths between 7.1 m and 7.6 m during the mid-depth tow survey. They were located along the northern edge of the ZID at closest-approach distances between 16.3 and 18.8 m from the diffuser structure. Within that portion of the plume, effluent had been diluted more than 400-fold.

eliminated in the screening procedure. Nevertheless, they demonstrate why apparently significant excursions in transmissivity can occur when BNLs are present.

The natural variability thresholds are a statistical construct specifically designed to identify highly unusual measurements within the database of observations measured at the time of a given survey. Ostensibly, if these highly unusual observations resulted from poorly dispersed effluent beyond the ZID, they could constitute an exceedance of receiving-water limits. However, while these outlier analyses provide a useful screening tool for the compliance determinations, further review as to the cause of the excursion is always warranted.

This is particularly true when natural processes create highly localized excursions of large amplitude that are under-represented in the database of ambient measurements used to determine the 95% confidence bounds that establish the thresholds in Table 3.10. BNLs are the natural process that most often produces these significant excursions. Because the layers are very thin, they are encountered in only those CTD measurements collected close to the seafloor and because of this, they are generally under-sampled in the database as a whole. Additionally, because BNL water clarity is often markedly reduced, the handful of BNL transmissivities contrast sharply with average conditions determined from the more numerous water-clarity measurements collected throughout the rest of the water column.

In fact, the large transmissivity reductions associated with BNL encounters that make it through the screening procedure lend credence to the overall approach for identifying measurements of potential compliance interest. Identification of these highly localized anomalies demonstrates the efficacy of statistical construct. If the ambient variability estimates were artificially large, or if the confidence thresholds were too lax, significant excursions in seawater properties of potential compliance concern could be missed. Instead, these examples demonstrate that the selection of the standard 95% confidence interval for compliance screening are capable of identifying anomalous excursions, even though a few false positives may result. In fact, false positives are expected from the very definition of the confidence level; namely, one-in-twenty apparently “significant” excursions would be expected to occur by chance alone when a 95% confidence level is applied in a null hypothesis test.

3.3.2 Numerical Limits

The last four permit provisions (P3 through P6 in Table 3.7) provide numerical limits that identify unacceptable changes in receiving-water properties. These provisions are derived from three regulatory documents: the Central Coast Basin Plan, the COP, and by reference, the California Thermal Plan. An additional quasi-quantitative threshold on Secchi depth measurements can be developed from the permit limit on the transmission of natural light (P4 in Table 3.7).

COP Objectives

The COP allows DO to be depressed 10% below “*that which occurs naturally*” due to the presence of oxygen-demanding material within effluent (COP §II.D.1; Permit Provision P5 in Table 3.7). Likewise, pH is allowed to extend 0.2 units beyond natural ranges (COP §II.D.2; Permit Provision P6 in Table 3.7). Numerical permit limits are therefore determined by adding the COP allowances to the thresholds of natural variability listed in Table 3.10. After applying these allowances, the permitted range in wastewater-induced excursions in DO and pH encompass a slightly broader range (*cf.*, bottom three rows of Table 3.10 and Table 3.11). However, because none of the 361 DO and pH measurements that were left after applying the first two screening questions in Table 3.9 exceeded the natural variability thresholds, all of these measurements were also in compliance with the less-restrictive COP receiving-water objectives. In fact, none of the 46,688 receiving-water measurements of DO and pH collected

during 2016 exceeded each survey's respective natural variability threshold, much less the less-stringent numerical limits listed in Table 3.11.

Table 3.11 COP Limits on Discharge-Related Changes to Seawater

Water Property	March	May	September	October
Temperature (°C)	>15.54	>15.95	>18.41	>17.02
Transmissivity (%)*	<57.1	<68.7	<70.8	<71.2
*Applicable Depth (m)	0-6.5	0-8.5	0-10	0-10.5
DO (mg/L)	<3.83	<4.94	<4.81	<4.86
pH (minimum)	<7.602	<7.642	<7.664	<7.623
pH (maximum)	>8.236	>8.393	>8.329	>8.257

Although the COP remains silent regarding allowable thermal change, it incorporates the California Thermal Plan (SWRCB 1972) requirements by reference (COP Introduction §C.3). The Thermal Plan restricts temperature increases caused by new discharges to coastal water to be less than 2.2°C (4°F). As with DO and pH, a quantitative permit limit on temperature increases can be established for each survey (Table 3.11) by combining the Thermal Plan allowance with the natural variability thresholds listed in the first row of Table 3.10. As with DO and pH, none of the 46,688 receiving-water temperature measurements collected during 2016 exceeded each survey's respective natural variability threshold, and thus, all of the measurements also easily complied with the Thermal Plan's limits. In fact, maximum temperatures measured within the rising effluent plume were uniformly below that of the surrounding seawater because cooler seawater near the seafloor had been entrained in the plume shortly after discharge. Consequently, any potential thermal impact resulting from the discharge of warm wastewater was almost immediately eliminated upon discharge because the effluent entrained much colder seawater.

The COP objective limiting significant reductions in the transmission of natural light can also be translated into a numerical permit limit. Because the COP does not specify an allowance beyond natural conditions, the permitted minimum transmissivities (second row of Table 3.11) match the lower bound of the natural transmissivity range applicable to each survey (second row of Table 3.10). However, as described in the previous section, the transmissivity limits only pertain to a portion of the water column. Specifically, the penetration of ambient sunlight is largely restricted to the euphotic zone, which extends to twice the Secchi depth. Thus, the quantitative limits on transmissivity reductions only apply within the depth range of the euphotic-zone listed after the asterisk in Table 3.11. The 22 low transmissivity observations identified as being of potential compliance interest (Table 3.9) were all collected at depths exceeding 7 m and thus, were beneath the 6.5 m base of the euphotic zone (March applicable depth in of Table 3.11).

In addition to determining the extent of the euphotic zone, Secchi depth measurements provide additional measurements capable of assessing potential discharge-related impacts to natural light penetration. However, the Secchi depth's measure of light transmission is far less precise than the instrumentally recorded transmissivities used in the screening analysis. Secchi depths are determined visually, by lowering a standard-sized disk into the water column, and noting the depth at which it disappears. Because of the somewhat subjective nature of the measurement, Secchi depths are generally only reported to the nearest 0.25 m. In addition, artificial differences in the reported depth among stations can be introduced, for example, from changes in ambient light arising from changes in cloud cover as the survey progresses, or from collecting measurements inside or outside of the survey vessel's shadow.

Nevertheless, statistical hypothesis tests can be applied to the measurements to determine whether there are potentially significant reductions in average Secchi depths measured close to the diffuser structure

(Table 3.12). For each of the four surveys, the tests found that any perceived reduction in the transmission of natural light was too small to be of statistical significance at the 95% confidence level. Although the average nearfield Secchi depths were slightly less than the reference-station average during three of the four surveys (Table 3.12), the differences were too small to be resolved given the inherent measurement error in the Secchi depth measurements.

Table 3.12 Average Secchi Depths (m) measured near the Diffuser and at Distant Locations

Survey	Nearfield ¹	Reference
March	3.0	3.25
May	4.0	4.25
September	5.0	5.0
October	4.75	5.25

Specifically, a one-tailed two-sample t-test assuming equal variances found no statistically significant reduction in the nearfield mean depths at the 95% confidence level ($p > 0.07$).² The 0.25 m reduction found during the March and May surveys equaled the reporting precision for the Secchi depth measurements; accordingly, the $p = 0.16$ significance of the difference was well above the 95% threshold ($\alpha = 0.05$). The $p = 0.07$ significance of the 0.5 m difference in averages during the October survey, however, was marginal, and was largely driven by an unusually shallow, 4.5 m Secchi depth measured at Station RW3. That measurement was significantly shallower ($p = 0.03$) than the average depth measured at the other stations. Not surprisingly, that shallow Secchi-depth was measured well within the effluent plume as it rose to the sea surface within the ZID. At that location, the permit limitation on light transmission did not apply. Moreover, the markedly decreased transmissivity within the plume at that location resulted from entrainment of turbid ambient seawater that was present within the BNL near the seafloor during the survey. It was not the result of particulate loading that was present within the effluent itself.

Basin Plan Limits

The COP limits, which are benchmarked to “*that which occurs naturally*,” allow adaption to seasonal and interannual changes in the ambient marine environment surrounding the outfall. In contrast, the Basin Plan contains fixed numerical limits restricting DO concentrations to no less than 5 mg/L (P5 in Table 3.7), and pH levels to a range of 7.0-to-8.3 pH units (P6). As such, the Basin Plan’s inflexible DO and pH limits fail to account for natural oceanographic processes, like upwelling, that may result in excursions in ambient seawater that range well beyond those limits.

Although none of the 2016 receiving-water observations ranged beyond the Basin Plan limits on DO and pH, the lowest measured DO of 5.61 approached the Plan’s 5-mg/L lower-bound limit, and the highest measured pH of 8.102 approached the Plan’s 8.3 upper-bound limit. Both of these extrema were measured within ambient seawater and well away from the influence of the discharge. These, and other past DO and pH measurements of ambient seawater within the survey area call into question the applicability of the fixed Basin Plan limits to ocean waters. In fact, published range-acceptability criteria that are used to assess the validity of CTD data in this and other California regional monitoring programs characterize seawater with DO concentrations as low 3 mg/L and with pH as high as 8.5 as being reasonable along the California mainland shelf.³ Not surprisingly, DO and pH concentrations measured within ambient seawater during past MBCSD surveys, have ranged well beyond the Basin-Plan limits.

For example, some or all of the measurements collected during the March and April 2010 surveys, the August 2011 survey, and the May 2013 survey significantly exceeded the Basin Plan’s upper bound pH

¹ within 37 m of the diffuser structure

² As defined in the acronym list at the beginning of this report, the “*p*-value” determines the significance of any perceived difference in mean Secchi depths by testing it against a null hypothesis of no difference in means. It quantifies the probability that a perceived difference in mean depths could have occurred by chance. When $p < 0.05$, there is less than a 5% risk that the observed spatial gradient was happenstance due to random fluctuations.

³ Table 3 in the field operations manual for the Southern California Bight Study (SCBFMC 2002)

threshold of 8.3 units (MRS 2011, 2012, and 2013). In those cases, perfunctory application of the Basin Plan's pH threshold could have led to the incorrect conclusion that the discharge had caused an unacceptable increase in pH. In reality, the pH levels were the result of regional phenomena that were not particularly unusual within ambient seawater along the California coast.

As with pH, DO concentrations within ambient seawater surrounding the outfall also occasionally fall outside the boundaries considered acceptable by the Basin Plan. Such was the case during the May 2014 survey when 266 observations ranged below the minimum Basin Plan limit of 5 mg/l (MRS 2016a). As with the pH measurements, these DO observations were well within the range considered typical of ambient waters at the time of that survey, and therefore complied with the COP portion of the NPDES permit provisions. The low DO concentrations arose because the deep watermass that was transported shoreward and into the survey area by upwelling was naturally depleted in oxygen. DO concentrations below 5 mg/L have been observed in a number of past water-quality surveys in conjunction with prolonged upwelling events (MRS 2011, 2012, 2013, and 2014). On a statistical basis, natural variability in DO concentrations during March 2016 surveys could be expected to range as low as 4.26 mg/L (Table 3.10), which is well below the 5 mg/L Basin Plan limit.

Clearly, DO and pH variations beyond their respective fixed limits were simply not envisioned for coastal ocean waters when the Basin Plan was promulgated in 1972. The fixed Basin Plan limits were largely designed for discharges to onshore surface waters (e.g. streams and lakes), where little natural variation in DO and pH is expected. In contrast to the Basin Plan limits, the COP recognizes the potential for inherent variation in the receiving-water characteristics, and specifies limits on excursions in these two water properties relative to background levels present at the time of the survey. Because the COP receiving-water objectives are designed to be adequately protective of the marine environment, application of the fixed Basin Plan limits to the same receiving-water characteristics already covered by the COP is not only redundant but also inappropriate. For these reasons, this annual report as well as past monitoring reports (MRS 2011, 2012, 2013, 2014, 2015, and 2016a) have recommended removal of the Basin Plan limits from future MBCSD discharge permits (see the recommendation in Section 5.4).

The Basin Plan also limits temperature excursions to those that will not adversely affect beneficial uses (P3 in Table 3.7). As documented by the response to Question 3 in Table 3.9, however, the presence of dilute wastewater particulates beyond the ZID exerted no perceptible thermal influence during 2016 and thus, could not have affected any of the beneficial uses designated for the receiving waters surrounding the outfall.

3.3.3 Other Lines of Evidence

The screening analysis described in Section 3.3.1 serves to restrict attention to the oceanic area where the permit provisions apply, and to eliminate excursions that were either unrelated to the discharge, or too small to be of consequence given the natural variability in the receiving waters. As a result, nearly all of the 2016 observations were summarily eliminated from further compliance analysis. However, within the eliminated measurements lies a body of additional evidence that further substantiates the finding that the receiving-water measurements recorded during the 2016 surveys complied with the limitations specified in the NPDES discharge permit. Three of these major other lines of evidence are described below: the high mixing performance of the outfall, the low discharge volume, and the high quality of the effluent that was discharged.

Abductive inference (Suter 2007) is a procedure for analyzing these varied lines of evidence. It emphasizes a pattern of reasoning which accounts for both the discrepancies among multiple lines of evidence as well as concurrences. It has been used to implement sediment-quality guidelines for California estuaries (SWRCB 2009). Application of this “*best explanation*” approach to the water quality dataset serves to limit the uncertainty associated with each individual CTD measurement and provide a more robust compliance assessment. Together, these lines of evidence significantly strengthen the conclusion that the discharge fully complied with the NPDES permit during 2016. In particular, it supports the finding that the localized presence of dilute wastewater constituents had no tangible effect on the ocean environment surrounding the outfall.

Mixing Performance

The high-spatial-resolution tow measurements recorded during 2016 demonstrated that even well within the ZID, wastewater was being diluted to levels far higher than expected from the dilution modeling. Although the screening procedure eliminates observations within the ZID because the permit limits do not apply there, these close-in observations foretell whether the outfall is capable of routinely meeting the 133-fold dilution necessary to achieve COP objectives that apply after completion of the initial dilution process.

Some of the lowest dilutions measured during 2016 were recorded during the May survey, only 3.3 m from the southeastern end of the diffuser structure (red shading Figure 3.9). Those measurements documented the presence of wastewater that had been diluted more than 266-fold shortly after discharge and long before the initial dilution process was complete (MRS 2016d). Nevertheless, the measured dilution was double the 133:1 critical initial dilution used to establish limits on contaminant concentrations in MBCSD wastewater.

In addition, this minimum dilution was measured well within the ZID, near a diffuser port, and at a depth of 8.6 m, which was 2.2 m deeper than the 6.4-m trapping depth identified in the modeling that established the 133:1 critical initial dilution ratio. Based on its location, the initial dilution process had only just begun and, according to the conservative modeling results, dilution levels would be expected to be substantially less than 133:1 at that depth level. Instead, the 266-fold dilution measured shortly after discharge demonstrates that the diffuser structure was dispersing the effluent far more efficiently than predicted by the conservative initial-dilution model.

These measurements, and the large number of other measurements collected close to the diffuser structure during 2016 demonstrate that the COP receiving-water objectives were being routinely met by the limits on chemical concentrations within discharged wastewater that are promulgated by the NPDES discharge permit issued to the MBCSD. This includes measurements collected well within the ZID and before completion of initial mixing. Measured dilution levels throughout the year easily exceeded the 133:1 critical initial dilution used to establish end-of-pipe permit limitations on contaminant concentrations within wastewater discharged from the MBCSD treatment plant.

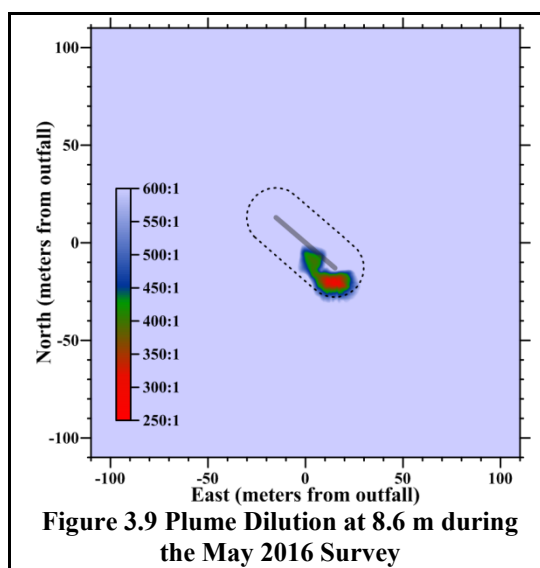


Figure 3.9 Plume Dilution at 8.6 m during the May 2016 Survey

In addition, only slight salinity and density anomalies showed any indication of the presence of dilute wastewater constituents within the survey area during 2016. The absence of wastewater-induced signatures in other seawater properties was the direct result of the high dilution achieved within a few meters of a diffuser port, and the low contrast between effluent and seawater properties other than salinity. This supports the conclusion that changes to receiving-water quality caused by the discharge of MBCSD effluent was imperceptible against the backdrop of natural variation in temperature, transmissivity, DO, and pH.

Limited Discharge Volume

In addition to high dilution, the small volume of wastewater discharge further clarifies why wastewater constituents do not exhibit a perceptible influence on seawater properties, even shortly after discharge. As such, the limited discharge volume provides another line of evidence supporting the conclusion that the MBCSD discharge complied with permit limits designed to protect the beneficial uses of receiving waters.

Largely because of the small discharge volume, plume signatures observed during receiving-water surveys have been of limited lateral extent, and tended to be highly localized within the ZID surrounding the diffuser structure (dotted line in Figure 3.9). As discussed in Section 3.2.1, the reason for the small discharge footprint can be appreciated when the low MBCSD discharge rate is evaluated relative to the volume of seawater encompassed within ZID. At the 0.843 MGD average discharge rate in 2016, an entire day of discharge would only displace 7% of the seawater within the ZID, and it would take 14 days to fill the entire ZID volume. The MBCSD discharge's tiny fraction of the receiving-water volume pales in comparison to that of the vast majority of other municipal discharges to US coastal waters, some of which have discharge rates 200-times larger than that of the MBCSD WWTP.

In addition, because of the combined effect of low discharge rate and rapid dilution, plume measurements within the MBCSD survey area are almost always exclusively delineated by reductions in salinity alone. Excursions in other water properties caused by the presence of dilute wastewater constituents have never been observed except extremely close to the diffuser structure and within the ejection jet from an individual diffuser port. Although 5,722 CTD measurements were collected within the ZID during 2016, none was close enough to a diffuser port to discern the influence of wastewater constituents, other than by a reduction in salinity. Although anomalies in some seawater properties were apparent in the upper water column in conjunction with the plume's low salinity, they arose from the entrainment and upward displacement of ambient seawater rather than from the presence of wastewater constituents.

However, in 2015 (MRS 2016a), a set of CTD measurements were collected within an ejection jet and provided a rare glimpse into the direct influence of wastewater constituents on water-property measurements. Small but perceptible increases in temperature, and reductions density, DO, pH, and transmissivity were captured as the CTD passed within 22 cm of a diffuser port. These coincided with the largest-amplitude salinity anomaly ever encountered in the MBCSD receiving-water quality database, and the associated 48:1 dilution was far lower than any previously encountered. These measurements demonstrated that the influence wastewater constituents on receiving-water properties, other than salinity, begins to disappear at a dilution close to 48-fold, and at a distance of less than 0.5 m from each diffuser port. Thus, the perceptible influence of the comparatively low-volume MBCSD discharge is restricted to a small region around each diffuser port, and does not materially comele with the discharge from adjacent ports, as is often the case with larger volume dischargers.

Effluent Quality

The consistently high quality of the MBCSD effluent provides another independent line of evidence that demonstrates why significant water-quality impacts would not be expected from the MBCSD discharge. Specifically, the high treatment level achieved by the MBCSD WWTP results in the removal of nearly all (90.0%) of the suspended solids and oxygen-demanding material (82.9%) from the wastestream prior to discharge (Table 2.2). The small residual amount of remaining material is then easily dispersed by the high-efficiency diffuser structure that discharges into the particularly well flushed, open-ocean environment within Estero Bay. As a result, the discharge would not be expected to contribute materially to turbidity within the dilute effluent plume; or to excursions in DO, pH, and temperature.

For example, oxygen-demanding material discharged by the MBCSD WWTP does not accumulate within seafloor sediments because the dispersive capacity of the energetic open-ocean environment of Estero Bay prevents wastewater particulates from settling near the outfall. Within the water column, organic-decay modeling demonstrates that the DO demand (BOD) within the MBCSD effluent would be less than 3 mg/L upon discharge (Page II-8 of MRS 2003a). After mixing 133-fold, the dilute BOD loading would induce a minuscule DO depression of no more than 0.022 mg/L. Because similar levels of BOD loading are typical of most ocean dischargers, the National Academy of Sciences (Page 9; 1993) determined that DO depletion from the discharge of municipal effluent is “*not of ecological concern in the ocean or open coastal waters,*” and when it is of concern, such as in estuaries, it is “*more likely to result from eutrophication by nutrients rather than point source inputs of BOD*” (Page 9 of 1993).

The low potential for water-quality impacts from the MBCSD suspended-solids discharge can be demonstrated in a similar manner. As with DO, wastewater-induced turbidity is imperceptible except within a diffuser’s ejection jet. This is the case because of a combination of three factors, the low suspended-solid concentrations within the MBCSD effluent, the high dilution levels achieved shortly after discharge, and the large natural variability in the transmissivity of ambient seawater. This can be demonstrated quantitatively for each survey by projecting the transmissivity decrease that would result from the discharge of effluent with a TSS concentration equal to that measured onshore on the day of the survey (Table 3.13). After completion of the initial dilution process, the projected reduction in measured transmissivity due to effluent solids loading is projected to be 1% or less. This potential influence on receiving-water turbidity measurements is small compared to the observed range in ambient transmissivity, which, during the 2016 surveys, was at least 18-times greater.

Table 3.13 Projected Transmissivity Reduction caused by Wastewater Particulates and the Background Transmissivity Range

Parameter	March	May	September	October
Initial Dilution	534	284	512	489
Effluent TSS (mg/L)	22.0	25.6	46.0	69.0
Transmissivity (%)				
Projected Reduction	0.2	0.6	0.7	1.0
Natural Range ¹	47.2	12.4	12.8	24.9

¹ Observed difference between the most extreme transmissivity measurements collected beyond the influence of the discharge

3.3.4 Visual Observations

Two remaining permit provisions (P1 and P2 in Table 3.7) are not addressed by the foregoing analysis of instrumental observations. These provisions assess whether floating particulates, oil and grease, or any aesthetically undesirable discoloration of the ocean surface was present during the receiving-water surveys conducted in 2016. These permit provisions are instead addressed by visual observations made during each of the surveys. The visual observations documented during the 2016 surveys found no evidence of these discharge-related impacts to the ocean surface.

3.4 BENEFICIAL USE

Additional visual observations compiled from the 2016 receiving-water surveys demonstrated that the coastal waters near the outfall and along adjacent shoreline continued to be beneficially used by humans and wildlife. There was no evidence indicating that the discharge detracted from the beneficial uses that were identified for northern Estero Bay in the Basin Plan (see Section 2.1.2). Specifically, fish, marine mammals, and seabirds were observed utilizing the area within and around the ZID area as marine habitat (*viz.*, the *MAR* beneficial-use identified for the waters of Estero Bay in the Central-Coast Basin Plan, as outlined in Section 2.1.2). The presence of pedestrians, equestrians, sailboats, and commercial and recreational fishing boats attested to the use of the waters for water recreation (*REC-1* and *REC-2*), transportation (*NAV*), and fishing (*COMM*).

4.0 MARINE SEDIMENTS AND BENTHIC BIOTA

This chapter describes the coastal benthos within Estero Bay where the MBCSD effluent is discharged. The MBCSD Offshore Monitoring Program includes benthic observations to assess potential impacts of wastewater discharge on marine sediments and the infaunal community near the outfall. Benthic environments are important indicators of the presence of marine pollution because they are the major reservoir for most contaminants that enter the ocean. Deleterious changes in the benthic environment, resulting from the deposition of effluent particulates, have been observed around other coastal outfalls (Stull et al. 1986a); however, no such impacts have been found after three decades of benthic monitoring around the MBCSD outfall. This includes the October 2016 survey described in this chapter.

In the marine ecosystem, organic and inorganic matter are dynamically interrelated, and the quality of the material deposited on the seafloor directly affects the health of infaunal communities. Therefore, both the physicochemical and biological properties of the benthos are described in this chapter. Infauna residing within sedimentary environments serve as indicators of contaminated sediments because of their limited mobility and well-defined responses to pollution. Numbers of species, abundance, biomass, and other parameters of community composition can indicate contaminant-caused stresses if, for example, gradients are observed extending from a pollutant source to more distant, unaffected areas.

This chapter presents the full results of the benthic survey conducted on 25 October 2016. Physical properties of sediments are tabulated in Appendix C and laboratory chemistry results from the survey are provided in Appendix E of this report, while benthic biological data are provided in Appendix B. BC Laboratories conducted the relevant chemical analyses following methods described by Tetra Tech (1986ab). Marine Research Specialists performed the infaunal community analyses and data synthesis.

The following section describes the benthic-monitoring program, including collection methods and station design. The second section of this chapter presents the findings related to the physicochemical analysis of the sediments. The third section describes the health of the benthic infaunal community residing within Estero Bay during 2016.

4.1 MONITORING PROGRAM

As with the marine receiving-water monitoring described in Chapter 3, sediment samples were collected at reference and outfall stations to investigate any potential discharge-related impacts to the benthos. The samples were analyzed for physicochemical properties and infaunal community composition in order to compare the benthic environment at these two sets of stations, as well as at gradient stations in between.

4.1.1 Scope of Monitoring

On 25 October 2016, offshore sediment samples were collected at seven stations within Estero Bay (Figure 4.1). The field sampling and laboratory procedures adhered to the requirements of the Monitoring and Reporting Program described in the discharge permit (RWQCB-USEPA 2009). During the three decades of offshore monitoring, the benthic-monitoring program has undergone a number of major modifications. The original permit, issued in 1985, required semiannual sediment collection at six stations with physicochemical analysis of a composite of three replicate grab samples collected at each station. The stations were distributed around the outfall along a north-south isobath. Five of these original stations have continued to be sampled throughout the monitoring program. The sixth station, a “reference” station, located 1 km to the south, was dropped when the permit was reissued in 1993 because it was found to be unrepresentative of the benthic environment near the outfall.

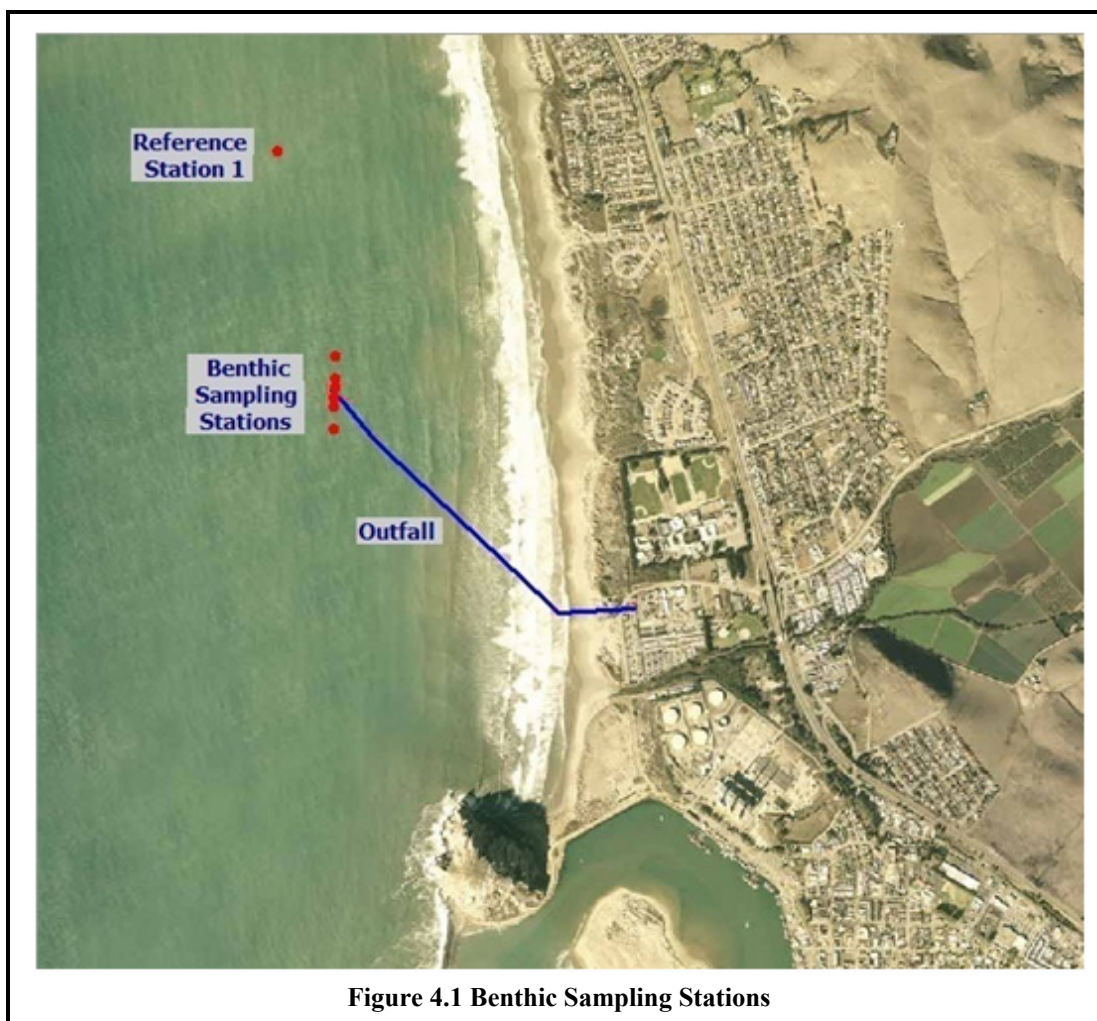


Figure 4.1 Benthic Sampling Stations

The discharge permit that was reissued in 1993 (RWQCB-USEPA 1993ab) also changed laboratory protocols. It required separate chemical analysis of each of the replicate samples collected at the five Stations. Statistical analysis of the scatter in chemical concentrations measured within the replicate samples at each individual station quantified the within-station variability and quantified the power of the monitoring program to detect spatial differences among stations. In particular, this report uses the replicate chemical analyses conducted in 1993 and 1994 to establish confidence intervals around concentrations measured at each station. If the observed differences in chemical concentrations among stations are less than the inherent uncertainty in the concentration at a given station, as defined by variability in replicate samples, then the spatial differences between stations cannot be considered statistically meaningful. In 1995, after the within-station variability had been quantified, chemical analysis reverted to a composite protocol where aliquots from each of the three replicates collected from a station were homogenized into a single composite sample in the laboratory before extraction.

Further modifications were implemented when the NPDES permit was reissued in 1998 (RWQCB-EPA 1998ab). After 13 years of monitoring, the amplitude of seasonal fluctuations had been adequately determined, eliminating the need for a post-winter survey. As a result, the focus of monitoring shifted to the determination of interannual changes in spatial patterns. To that end, four additional benthic sampling

stations, located 60 m from the outfall, were added to the five historical stations. The cross-shore configuration of two of these new stations (Stations 8 and 9) provided a more spatially balanced sampling pattern capable of detecting potential impacts that might not be restricted to along-shore directions. Finally, chemical analyses were performed on a single grab sample collected at each of the nine stations.

Based on an analysis of the previous 23 years of benthic data, the current permit (RWQCB-EPA 2009), adopted in January 2009, also instituted several changes to the program. First, chemical analysis protocols reverted to those utilized prior to 1998, where aliquots from each of three replicates collected at a station are homogenized into a single composite sample prior to analysis. This reduces uncertainty in the reported station concentrations by reducing within-station variability. Second, the permit eliminated the analyses of sulfides within benthic porewater samples. High-resolution analyses conducted over the previous six years had definitively demonstrated that the Estero Bay sediments do not contain measureable levels of dissolved sulfides. Finally, the cross-shore benthic monitoring stations (Stations 8 and 9) were eliminated from the sampling pattern because inherent differences in the benthic environments caused by their difference in water depth confounded the interpretation of trends at the remaining stations.

The current permit also eliminated the sampling requirement at the northern reference station (Station B1); because of its physical separation (1 km) from the outfall, the reference station has a measurably different infaunal community composition (MRS 1998bi) and sediment physicochemistry from the remaining stations. However, as one of the six original sampling stations in the program, the long record of grain size, infaunal, and metal-concentration data acquired at Station B1 has immeasurable value for the insight it provides into the larger scale processes working within Estero Bay as a whole. Therefore, sampling for sediment parameters at Station B1 has been voluntarily continued.

4.1.2 Sampling Station Design

Benthic sampling during 2016 was conducted at the seven stations shown in Figure 4.1 on Page 4-2 and in Figure 4.2 on Page 4-6. The target positions for sample collection are specified in the current NPDES permit and are summarized in Table 4.1. The stations lie along a north-south isobath (15.2 m) that intersects the center of the diffuser structure. Two stations (B4 and B5) are located within 20 m of the center of the diffuser structure, at the edge of the “zone of initial dilution” (ZID). The locations of these two stations coincide with those of the receiving-water Stations RW3 and RW4 (Figure 3.2). Two additional stations (B3 and B6) are located at nearfield distances (60 m) from the diffuser structure and coincide with Stations RW2 and RW5. Finally, the midfield stations (B2 and B7) are separated by three diffuser lengths from the discharge (150 m). The seventh, reference station (B1) is located 1 km north of the outfall, well beyond the discharge’s sphere of influence.

Table 4.1 Target Locations of Benthic-Sediment Sampling Stations

Station	Description	Latitude	Longitude	Distance (m)	
				Closest ¹	Center ²
B1	Upcoast Reference	35° 23.730' N	120° 52.677' W	1,000	1,016
B2	Upcoast Midfield	35° 23.280' N	120° 52.504' W	138	150
B3	Upcoast Nearfield	35° 23.231' N	120° 52.504' W	49	60
B4	Upcoast ZID	35° 23.210' N	120° 52.504' W	15	20
B5	Downcoast ZID	35° 23.188' N	120° 52.504' W	15	20
B6	Downcoast Nearfield	35° 23.167' N	120° 52.504' W	49	60
B7	Downcoast Midfield	35° 23.118' N	120° 52.504' W	138	150

¹ Distance from the closest open diffuser port

² Distance to the center of the open diffuser section

As described in Section 3.1.3, the finite length of the diffuser structure creates ambiguity in the distance measured from the outfall. For discussion purposes, station separations are usually referenced to the center of the diffuser. However, in terms of contaminant dispersion, the closest-approach distance is the controlling parameter.

The upcoast reference station (B1), midfield stations (B2 and B7), and ZID stations (B4 and B5) have all been regularly sampled throughout the 31 years of benthic monitoring. The two nearfield stations (B3 and B6) were incorporated into the program in 1999, and improved the program's ability to resolve potential benthic impacts related to effluent discharge. Although sampling is no longer required at Station B1 under the current permit, samples were voluntarily collected at the station and analyzed for grain size, infaunal composition, and trace-metal concentrations.

The extraordinarily long record of infauna, grain size, and trace metals has provided valuable insights into previously unknown trends in Estero Bay's regional benthic conditions. Therefore, although these trends are unrelated to the MBCSD discharge, the lengthy time history at Station B1 was deemed of enough scientific value to warrant continued sampling there. However, although required for compliance evaluation at the other stations, sediment oil and grease concentrations have not been found to be diagnostic of the seafloor environment, or the effects of the MBCSD discharge, so the voluntary sampling at Station B1 did not include those constituents.

4.1.3 Benthic Sample Collection and Processing

Field sampling and laboratory procedures adhered to the requirements of MBCSD Monitoring and Reporting specified in the permit (RWQCB-USEPA 2009). The 38-ft F/V *Bonnie Marietta*, owned and operated by Mark M. Tognazzini of Morro Bay, provided vessel support during the offshore survey. Dr. Douglas Coats of Marine Research Specialists (MRS) logged navigation fixes, and collected sediment subsamples for chemical assay. Cletis England, a biologist with MRS, sieved and processed infaunal samples. Dean Dusette, a compliance-monitor also with MRS assisted in benthic grab deployment and retrieval, in conjunction with Vessel Marine-Equipment Technician William Skok, who was responsible for overall deck operations.

Equipment

Sediment chemistry and benthic infaunal samples were collected using a chain-rigged Young grab whose design was modified from that of a Van Veen sediment sampler. The pair of jaws on the grab acquires sediments from a 0.1-m² area of the seafloor. The stainless-steel grab is coated with Dykor[®] and is equipped with a frame that enhances its penetration, stability, and proficiency in collecting a level, undisturbed sample. Dykor[®] has properties similar to Teflon[®]; it improves the chemical inertness of the grab sampler, limiting contamination of the sediment chemistry samples by the grab itself, particularly from trace metals. The grab was completely refurbished by sandblasting and recoating with Dykor[®] in 1999. At that time, a stainless steel lip was also welded to the mouth of the jaw to improve the seal, which aids in sample retention during ascent.

At each of the seven monitoring stations, three replicate grab samples were collected and composited for analyses of sediment chemical composition and bulk properties, and five replicate grab samples were collected for benthic infaunal analyses (Appendix Table C.1).

Offshore navigation aboard the survey vessel was supplied by a Furuno[™] GPS 30 coupled to an FBX2 differential beacon receiver. Global positioning satellite (GPS) navigational fixes were recorded digitally at 1-second intervals. Electronic recordings of waypoints were used to mark the seafloor trip-time of the grab sampler and other pertinent information. Navigational errors inherent in standard GPS readings were

greatly reduced with the use of a Differential GPS (DGPS) system that was first implemented by the U.S. Coast Guard. DGPS incorporates a second signal from a nearby land-based beacon. Because the beacon is fixed at a known location, the position error in the reading from the GPS satellites can be precisely calculated. This correction is continuously transmitted to the DGPS receiver and results in extremely accurate offshore navigation, typically with position errors of less than 2 m.

Physicochemical Samples

Prior to and during the survey, the sampling equipment was thoroughly cleaned to eliminate the introduction of non-sediment contaminants into the samples, and to prevent cross-contamination between stations. Before the survey, the grab sampler and sediment scoops were washed with Alconox®, deionized water, and 10% Hydrochloric Acid (HCl). During the survey, the grab and sampling utensils were washed with Alconox®, rinsed with seawater, and subsequently decontaminated with deionized water, methanol, and 1% HCl prior to sampling at each station.

Surficial sediments were collected from the upper 2 cm of the single grab sample recovered from the seafloor at each of the seven benthic stations. Sufficient sample volumes were collected to provide material for QA/QC analyses. Samples were stored in the appropriate glass or plastic containers and refrigerated at approximately 4°C prior to analysis. Chain of custody forms accompanied all sample shipments from the field and between laboratories. Appendix E¹ contains the chains of custody, laboratory QA/QC analysis, and the chemical results as reported by the laboratory, while Appendix C includes a physical description of the samples.

The sediments from each of the seven benthic-monitoring stations were separately analyzed for the analytes listed in Table 4.2. A summary of the analytical methods used and their authority is provided in Table E.1.

Table 4.2. List of Analytes in Sediment Samples

- | | | |
|--------------------------|-------------------------------|---------------------------------|
| • Sediment particle size | • Biochemical oxygen demand | • Total Kjeldahl nitrogen (TKN) |
| • Moisture content | • Oil and grease ² | • Total volatile solids (TVS) |
| • 11 trace metals | | |

Infaunal Samples

In addition to samples collected for sediment chemistry, five replicate grab samples were collected at each of seven benthic-monitoring stations for infaunal analyses. Upon retrieval of the grab, each sample was inspected for acceptability. Acceptance criteria are based on penetration depth, surface condition, and overall sample integrity and are reported in Table C.1 in Appendix C of this report. Sediment samples that met the acceptability criteria were lightly washed, and elutriated onto a 1.0-mm mesh sieve. The extracted material was then washed into a labeled 16-oz. jar and preserved with 10% buffered formalin. After a minimum fixation period of 48 hours, the formalin was rinsed from the samples on a 0.5-mm mesh sieve and the samples were transferred to 70% alcohol for processing, preservation, and storage. After transfer to alcohol, the infaunal samples were stained with a Rose Bengal solution to aid in sorting of the organisms into the following major taxonomic groups: annelida and nemertea, mollusca, crustacea, echinodermata, and miscellaneous phyla.

All individual organisms were enumerated and identified to species level where possible. Voucher collections have aided in the consistent identification of organisms collected throughout the 31-year

¹ Note that the chemistry results reported by the laboratory in Appendix E for Stations B1 and B7 were reversed due to a labeling error in the field. Results reported elsewhere in this document have been corrected for this error.

² As discussed previously, the sediments from the voluntary sampling conducted at B1 were not analyzed for oil and grease.

benthic-monitoring program. The voucher collections contain specimens representative of each species identified, and are augmented as new taxa are collected. The qualifications and experience of the taxonomic specialists responsible for species identification are presented at the end of Appendix B.

Sampling Location Offsets

Figure 4.2 shows the position of the chemistry (blue) and infaunal (green) grabs, relative to each target location (red) that were recorded during the October survey. As described in Chapter 3, a number of factors influence the offset of the actual grab locations from the target locations listed in Table 4.1. Because the vessel is anchored during sediment grab sampling, differences between the sampling and target positions arise because the vessel swings about the anchor location along a scope of approximately 20 m. How the vessel will set relative to its anchor position is difficult to predict based on the constantly changing wind, wave, and current conditions. Changes in position during sampling at an individual station arise because of these changing conditions, in addition to movement from anchor drag. The within-station position changes during the October 2016 survey are apparent in Figure 4.2 by the spatial dispersion among the three sediment-chemistry grab samples and five infaunal grab samples collected at each station.

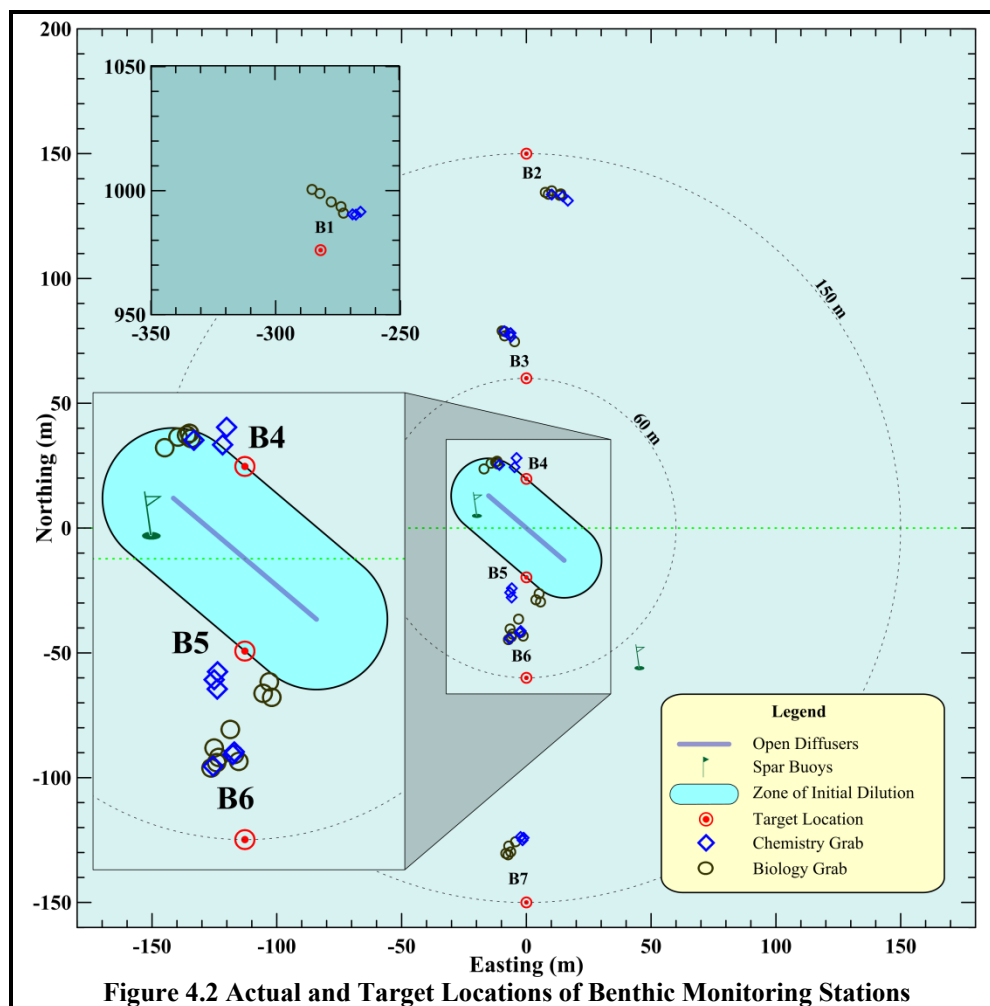


Table 4.3 lists the average position of the eight grabs collected at each station. Although it is not possible to collect samples at the precise target locations, knowledge of the actual location where samples were collected allows the spatial offsets to be factored into the gradient analysis. Without the benefit of differential GPS navigation, most of these spatial offsets would never have been resolved. The enhanced resolution provided by differential GPS even lends insight into minute spatial differences between individual grab samples that would not have been resolved otherwise.

Table 4.3 Average Location of Benthic-Sediment Grab Samples at each Station

Station	Description	Latitude	Longitude	Distance (m)		
				Closest ¹	Center ²	Offset ³
B1	Upcoast Reference	35° 23.737' N	120° 52.685' W	1015	1031	20
B2	Upcoast Midfield	35° 23.271' N	120° 52.496' W	124	134	20
B3	Upcoast Nearfield	35° 23.241' N	120° 52.509' W	65	78	19
B4	Upcoast ZID	35° 23.213' N	120° 52.511' W	14	28	12
B5	Downcoast ZID	35° 23.183' N	120° 52.505' W	24	30	10
B6	Downcoast Nearfield	35° 23.176' N	120° 52.507' W	36	43	18
B7	Downcoast Midfield	35° 23.130' N	120° 52.507' W	116	127	23

Despite the many unavoidable factors affecting vessel offset, differences between the target position and average grab location were only about one-third of the 30-m width of the ZID at the two ZID stations (B4 and B5). Vessel positioning at the ZID stations is particularly complicated by the potential for diffuser damage from the anchor and chain if deployment across the diffuser structure is required (See the inset in Figure 4.2). Slightly larger offsets occurred at more-distant stations to the north, including the Reference Station B1, where fidelity to the target locations is less important. Of particular note, wind speeds increased and shifted direction over the course of the survey, making the prediction of vessel offset relative to anchoring location more challenging as the sampling progressed from south to north.⁴

4.1.4 Benthic Analysis

A wide variety of physical, chemical, and biological parameters was determined from the 56 sediment samples collected during the October 2016 survey. This report examines these parameters for any potential discharge-related spatial patterns. They are also compared to historical measurements to reveal any potential temporal trends that could be associated with a buildup of contaminants. Finally, the interrelation between biological and physicochemical parameters is evaluated for possible toxicological effects.

Organic-Loading Properties

Sediment samples collected at the seven benthic-monitoring stations during the October 2016 survey were analyzed for moisture content, grain size, organic properties, and eleven trace metals. The organic-loading parameters, oil and grease (O&G), total volatile solids (TVS), total Kjeldahl nitrogen (TKN), and biochemical oxygen demand (BOD), are considered relevant for sewage-outfall monitoring. If the volume of organic solids discharged by the outfall is not adequately dispersed, then a buildup of O&G, TVS, TKN, and BOD can occur, and can impact areas beyond the ZID.

¹ Distance from the closest open diffuser port

² Distance to the center of the open diffuser section

³ Average offset from Target Locations listed in Table 4.1

⁴ Grab trip times are listed in Table C.1 of Appendix C.

BOD measures the depletion of dissolved oxygen caused by aerobic microorganisms during decomposition of organic material. Excessive BOD can be deleterious to marine organisms if decomposition significantly depletes dissolved oxygen levels. This can occur when aerobes metabolize excess organic matter deposited by wastewater discharge in a confined environment.

The TVS concentration quantifies the amount of organic matter in the solid fraction of bottom sediments. It is reported as percentage of dry-sample weight. TKN measures the amount of organic nitrogen present as ammonia, as well as that bound into organic compounds in the aqueous phase of the benthic samples. However, natural TKN levels vary widely among seafloor sediment samples, so anthropogenic (human-induced) impacts are difficult to discern from TKN measurements alone. Similarly, other properties that can indicate organic loading from effluent discharge, including dissolved sulfides, O&G, and BOD, are problematic to sample, analyze, and interpret.

For example, O&G concentrations are often below detection limits, even close to the discharge. Similarly, BOD is difficult to accurately measure in sediment samples, and interference from errant pieces of organic material, such as drifting kelp, has resulted in erroneously high levels at some stations in previous surveys. Specifically, Station B9 in 2000, and various stations prior to 1995 (Figure 4.9b on Page 4-26) exhibited anomalously elevated BOD concentrations. Because of persistent interference during BOD determinations, overall confidence in BOD results for benthic sediment samples remains low.

Trace Metals

In contrast to synthetic chemicals, the presence of trace metals within seafloor sediments is not necessarily indicative of anthropogenic input. Most trace metals are found in detectable concentrations within naturally occurring mineral deposits. Some metals provide essential minerals needed by marine organisms to survive. However, elevated levels of certain trace metals can be indicative of anthropogenic input, and excessive levels can cause deleterious effects in marine organisms.

Nine trace metals have been measured in seafloor sediments since the inception of the monitoring program. They are arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc. The toxicity of these metals to marine organisms has also been examined in numerous laboratory bioassays and field experiments (Long and Morgan 1991, Long et al. 1995). These studies have been summarized in the form of biological effects levels, most notably an effects-range low (ERL) concentration, below which toxic effects are not expected, and an effects-range median (ERM), above which adverse biological effects are anticipated. The potential for impacts from concentrations in the range between these thresholds cannot be reliably predicted.

In 1998, the analysis of aluminum and iron within benthic sediments was added to the monitoring program. Unlike the other nine metals, aluminum and iron are not particularly toxic to marine organisms, and their naturally occurring concentrations are relatively insensitive to anthropogenic input. This latter attribute makes them ideal tracers of natural variations in all the trace metals. Consequently, they have been used to normalize observed concentrations in the other nine metals so that background variation is reduced and anthropogenic trends can be revealed. In particular, iron has been used as a reference element for determining anthropogenic enrichment of trace metals within the Southern California Bight (Schiff and Weisberg 1997) and in many other offshore regions (e.g. Daskalakis and O'Conner 1995).

Physical Properties

Moisture content and grain-size distribution are two important physical properties of marine sediments. Moisture content is a measure of the volume of pore water present in the sediment samples. While not an indicator of discharge effects, it is an important parameter used to convert chemical concentrations measured in the wet sediment sample to a dry-weight basis. Dry-weight concentrations allow a more

direct comparison of chemical concentrations associated with bulk sediments by eliminating variations in sample mass that arise from arbitrary differences in water content. Moisture content is also used, along with salinity measurements, to eliminate the bias introduced by salt content in grain-size determinations.

Grain-size distribution is important to quantify because an increase in fine sediments near a municipal ocean outfall can be indicative of excessive effluent-solids deposition. In addition, the amount of fine sediment directly affects the composition of the infaunal community that resides within the sediments; although the precise mechanism for this relationship is rarely clear (Snelgrove and Butman 1994). Finally, natural variation in trace-metal concentrations has been correlated with the fine-sediment (mud) fraction and, like aluminum and iron, mud concentrations have been used to normalize metal concentrations (Dossis and Warren 1980, Ackerman et al. 1983, Horowitz and Elrick 1987).

Accurate measurement of the fine sediment fraction is particularly important for use in the interpretation of the marine biological data. Otherwise, variability in a benthic community's composition could be erroneously ascribed to effluent discharge, when instead it may be the result of natural variation in the distribution of fine particulates. One of the major difficulties in the accurate measurement of the fine fraction within marine sediments is that the standard pipette method described by Plumb (1981) and Folk (1980) overestimates the finest (clay) fraction. This occurs because the procedure incorrectly includes the weight of the dissolved solids (salt) when weighing the dried pipette extraction. This often generates an artificial bimodal grain-size distribution, where the smallest (clay) fraction exceeds the next largest (silt) fraction.

In 1994, the anomalous character of the bimodal size distribution in the MBCSD database led to a number of internal investigations and confirmatory grain-size analyses by outside laboratories (MRS 1995). The refined analysis method that is now in use accounts for the presence of dissolved solids by correcting for the measured salinity in the pipette fractions (MRS 1998i and Coats et al. 1999). This refined particle-size analysis method was instituted in the MBCSD monitoring program beginning in 1998. As a result, the finest size fractions determined in sediment samples collected after October 1997 are far more accurate than those determined in prior surveys. Therefore, some of the long-term trends in the MBCSD grain-size database can be ascribed to improvements in methodology, rather than actual changes in the benthic environment.

Biological Parameters

Determining whether the discharge is causing adverse biological effects involves assessing whether a balanced indigenous population (BIP) of shellfish, fish, and wildlife exists in areas potentially affected by the discharge. A BIP is an ecological community, which exhibits characteristics similar to those of nearby, healthy communities existing under comparable environmental conditions beyond the influence of the discharge. Certain biological characteristics of the community are examined in a BIP evaluation, including spatial and temporal distributions of species composition, abundance, biomass, dominance, and diversity. Other overall biological indicators of a BIP include trophic (feeding-habit) structure and the presence or absence of sensitive indicator species. Most marine monitoring programs determine the presence of a BIP using data on the infaunal organisms that live within sediments surrounding a potential pollutant source.

When assessing a BIP, infauna have many advantages over other marine organisms, such as marine mammals or finfish. First, infauna are relatively easy to collect in numbers large enough for reliable statistical testing. Second, they have limited mobility and cannot easily escape exposure to pollutants. Finally, decades of monitoring studies have established well-defined responses of infaunal taxa to pollutant stress.

In the earliest MBCSD monitoring surveys, analysis of biological samples produced two types of data on infaunal organisms: the number of organisms for each taxon, and the biomass of major taxonomic groups within each sample. Biomass is difficult to interpret, however, because the erratic presence of large, undersampled organisms severely skews the results. In addition, in practice, the heavy calcareous shells and exoskeletons of mollusks and crustaceans are troublesome to remove, and alcohol evaporation during weighing affects the determination of wet-weight biomass. Because of these problems, the USEPA does not recommend biomass for inclusion in 301(h) monitoring programs (USEPA 1987). Therefore, beginning in 1998, the NPDES permit no longer required this analysis.

Pollution affects marine ecosystems by changing the number and type of benthic organisms found in the sediments. However, subtle changes in community composition are not always readily apparent in the large volume of raw data generated by a long-term monitoring program. Over the 31 years of MBCSD benthic monitoring, 262,000 specimens representing 392 individual taxa have been collected. To assess whether the infaunal community has been impacted, these data must be summarized into concise parameters that are indicative of pollutant stresses.

Biodiversity is a common indicator of the well-being of ecological systems, and forms the cornerstone of most impact-assessment studies. Unfortunately, biodiversity is difficult to define quantitatively, in part because it has two major components: species richness and evenness. Species richness measures the variety (number) of species, while evenness measures the distribution of individuals (abundance) among the species (Magurran 1988). Healthy ecosystems are thought to be both rich in species and have an even distribution of individuals among those species.

Although no single measure can accurately represent changes in both evenness and richness along a pollution gradient, many indices attempt to do so. The Shannon Diversity Index (H') is the most widely used diversity index because it exhibits some degree of sensitivity to both evenness and richness. In addition, most diversity indices only measure the diversity within a local area at a particular time, *viz.*, among replicate samples from a single station (α -diversity). In contrast, pollutant stresses tend to induce marked changes in community composition between samples separated spatially and temporally (β -diversity; Smith et al. 1979). Multivariate analysis techniques are better suited to the evaluation of β -diversity than are individual indices.

Most diversity indices are also poorly suited for describing potential impacts to infaunal communities because they lack biological meaning, show little correlation with environmental quality, and are difficult to interpret ecologically (Hurlbert 1971, Goodman 1975, Washington 1984, Green 1979). As a result, ambiguous or biased estimates of diversity often occur. Therefore, many indices are not recommended for routine inclusion in 301(h) monitoring programs (USEPA 1987). Nevertheless, the current NPDES permit requires that a number of infaunal community indices be reported and examined for temporal trends (RWQCB-USEPA 2009). However, because of limitations associated with the indices, spatial differences with respect to outfall proximity, which are evaluated using standard statistical hypothesis tests, must be interpreted carefully. Table 4.4 lists the infaunal indices computed in this report. The associated algorithms and guidance on how to interpret them are presented in Appendix B.1.

Table 4.4 Infaunal Parameters Evaluated in the Monitoring Program

• Number of individuals • Number of individuals per species • Margalef species richness (d) (Margalef 1951) • Simpson dominance (C') (Simpson 1949, Wittaker 1965) • Pielou evenness index (J') (Pielou 1977) • Infaunal Trophic Index (ITI)	• Number of species • Species richness (S) (Magurran 1988) • Shannon-Wiener index (H') • Brillouin index (h) • Swartz dominance (S_w)
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4.2 MARINE SEDIMENT QUALITY

The quality of marine sediments surrounding the MBCSD outfall provides a definitive measure of potential marine impacts caused by wastewater discharge. Specifically, discharge-related impacts have been demonstrated in outfall monitoring programs offshore both central and southern California, wherein marked changes in the benthic environment have been observed surrounding the discharge locations (Phillips and Hershelman 1996, Stull et al. 1986b). Discharge-related impacts have also been observed on the continental shelf south of Estero Bay, where sediments were found to exhibit small but statistically significant chemical signatures indicative of drilling mud discharges over widespread areas (Hyland et al. 1994, Coats et al. 1991, Coats 1994). If the MBCSD effluent discharge were causing significant impacts to the marine environment, similar changes in sediment properties would be evident near the diffuser structure. The challenge is to distinguish changes that are caused by naturally occurring processes from those that result from human activities.

Physicochemical changes in sediments provide a good indication of marine impacts for four main reasons. First, because outfalls are located on the seafloor, sediments are proximal to the source of potential contamination. Second, many contaminants are hydrophobic and tend to preferentially adhere to the fine particulates as they settle out of the effluent plume. Third, contaminants incorporated into seafloor sediments tend to have a long residence time because of the slow dispersive processes that prevail within pore waters. Finally, infaunal organisms that live within seafloor sediments are continuously exposed to contaminants because they cannot easily escape the source of pollution. If seafloor sediments or pore waters become polluted, infaunal organisms can be impacted. Sedentary infaunal organisms provide a food source for other, more mobile organisms, such as finfish and shellfish. These trophic relationships can lead to bioaccumulation of contaminants within the marine food chain.

Despite the sensitivity of the benthos to effluent discharge, there has never been an indication of deleterious impacts to seafloor sediments resulting from the MBCSD discharge, including during 2016. The lack of negative benthic impacts is largely due to the comparatively low mass-emission of anthropogenic chemicals from the outfall. There has never been a large industrial contributor to the MBCSD influent stream, and the total discharge volume is small compared to ocean dischargers to the south, where discharge volumes can be over 100-times greater (SCCWRP 2006).

The lack of benthic impacts is evident from three sets of analyses performed on the chemical properties of sediment samples collected around the MBCSD outfall during the October 2016 survey. Each of these analyses is discussed in the following subsections. First, concentrations of potential contaminants measured in sediment samples were below levels that have been empirically linked to toxicity in marine organisms. Second, the spatial pattern of sediment concentrations near the outfall during 2016 bore no relation to outfall proximity. Third, sediment concentrations do not exhibit long-term trends that suggest a buildup of contaminants close to the outfall.

4.2.1 Concentrations below Toxic Levels

In keeping with the COP, narrative receiving-water limitations contained within MBCSD's NPDES discharge permit (RWQCB-USEPA 2009) state that the discharge should not cause chemical substances in marine sediments "*to be increased above levels which would degrade indigenous biota.*" This requirement was established because contaminants that reside in sediments for long periods have the potential to exert acute and chronic effects on resident marine organisms (NOAA 1991ab, Finney and Huh 1989, Bertine and Goldberg 1977).

Over the 31 years of continuous benthic monitoring, there has never been an indication of effluent-related degradation of marine organisms within Estero Bay. In large part, this is due to the consistently low chemical concentrations within the discharged effluent (Chapter 2), and the rapid dispersion effluent experiences shortly after discharge (Chapter 3). As a result, contaminant concentrations measured within Estero Bay sediments, both near and distant to the outfall, have been below thresholds reliably determined to be harmful to marine biota. In fact, the comparatively pristine marine environment of Estero Bay is reflected by benthic contaminant levels that are well below concentrations measured in the majority of sediments offshore southern California.

Thresholds of Biological Effect

Over the years, marine monitoring programs have placed considerable emphasis on benthic contamination. This experience has resulted in a set of thresholds of biological concern that can be used to screen chemical concentrations measured within marine sediment samples at any location. Specifically, these biological benchmarks can be used to assess whether the chemical concentrations measured in sediments have the potential for adverse biological effects. Although the thresholds are not regulatory criteria, they provide toxicological guidelines that help assess the potential environmental significance of specific chemical concentrations measured in the field.

The toxicological benchmarks advanced by NOAA (Long and Morgan 1991) are used in this report to evaluate sediment chemical concentrations measured during 2016. These guidelines are based on a correlation between chemical concentrations and observed biological effects in numerous modeling, laboratory, and field studies (Long et al. 1995). The studies identify two toxicological endpoints, an Effects-Range Low (ERL) and an Effects-Range Median (ERM) concentration. The lower toxicological endpoint (ERL) is associated with the concentration of a particular compound that produced an adverse biological effect in only 10% of the data. Below the ERL, adverse biological effects are unlikely. Above the ERM, which is based on the median of the toxicological data, adverse effects are considered likely to occur. The most conservative benchmark (ERL) was used in this report to determine the potential for marine biological impacts from all but one of the chemical concentrations that were measured within Estero Bay during 2016. These benchmarks provide a reliable screening tool for most, but not all sediment contaminants. For example, the relationship between nickel concentrations and the incidence of effects in the toxicological database is very weak. Consequently, even the upper (ERM) benchmark for nickel is a poor predictor of levels where biological effects might be expected.

Both trace-metals and organic contaminants are of concern in the benthic environment. However, their natural occurrence differs markedly. Synthetic compounds, such as PAHs, PCBs, and pesticides, are organic constituents that can have the most significant anthropogenic impact if found in sufficient concentrations. However, in this and many other marine sediment studies, their concentrations are below detection levels. In contrast, trace metals occur naturally in the marine environment and their concentrations are regularly detected in sediment samples. Because of this, trace metals serve as a powerful surrogate for evaluating the regional transport and fate of a host of other particulate contaminants.

Grain-Size Distribution

Within deep offshore basins, sediment trace metals accurately record the history of contaminant accumulation because they are ubiquitous and long-lived. However, these deep benthic sediments often contain significant fractions of fine-grained sediments, which support an increased number of trace-metal bonding sites per unit of volume. Thus, large differences in trace metal concentrations among sediment samples can be caused by natural differences in mud fractions that are unrelated to human activities. Because of this, many marine benthic studies divide (normalize) trace metal concentrations by the mud

fraction. Normalization is intended to reduce inherent natural variability, revealing anthropogenic contaminant patterns normally masked by the natural association between metals and fine particulates (Dossis and Warren 1980).

While this approach works well in deep quiescent basins, it is not applicable in nearshore environments where energetic oceanographic processes erode fine particles, leaving behind coarser, more transport-resistant sandy sediments. Sandy sediments usually have comparatively low metal concentrations. In addition, metals tend to be incorporated into the mineralogy of the sand grains themselves, rather than adsorbed onto their surfaces, as is the case with silts and clays. Therefore, metals are less likely to dissolve and become assimilated by marine organisms residing within sandy sediments. Because of the lower bioavailability of metals within sands, NOAA does not recommend mud-fraction normalization of trace-metals for samples with less than 20% fine-grained material (Long and Morgan 1991). Because the Estero Bay sediments consist almost entirely of sand (>98%; Table 4.5; Appendix Table C.3), iron- and aluminum-normalized concentrations are assessed instead.

Irrespective of trace-metal normalization, grain-size distributions can be indicative of a buildup of effluent particulates within the surficial sediments if, for example, there is an increase in fine fractions close to the discharge point. Additionally, the overall character of the grain-size distribution reflects the dynamics within a particular benthic environment. For example, higher mud fractions reflect more-quiescent depositional environments, while well-sorted sands result from energetic wave environments that winnow fine particulates as they constantly rework the surficial sediments. Accordingly, the predominance of sand-sized particulates in the grain-size distributions within the 2016 sediment samples reflects the high-energy environment near the outfall.

The sediment samples collected during October 2016 consisted of very-well-sorted³ fine sands with a distribution skewed slightly toward the finer fractions⁴ (Appendix Table C.4). During 2016, mean particle diameters ranged from 139 to 146 μm , lying in the middle of the fine-sand portion of the grain-size distribution. The high degree of sorting is reflected by low standard deviations ($0.352 < \sigma < 0.392$). The small, but consistently positive skew to the distribution reflects a slight excess in the finer fractions relative to larger fractions in the tails of the distributions. Together, these properties point to a highly energetic flow regime that tends to winnow fines from the surficial sediments. The contribution of larger-diameter fractions was negligible in that the combined coarse-sand and gravel fractions ($\leq 1\phi$) represented less than 0.58% of the total sample weight, except within the sample collected at Station B3, which

Table 4.5 Grain-Size Distribution (%) within Sea-floor Sediment Samples Collected on 25 October 2016

	Clay (9 to 12 ϕ)	Silt (5 to 8 ϕ)	Fine to Medium Sand (1 to 4 ϕ)	Coarse Sand (1 ϕ)	Very Coarse Sand (0 ϕ)	Gravel (-1 ϕ to -3 ϕ)
Station B1	0.50	1.24	97.99	0.09	0.07	0.11
Station B1 ¹	0.62	1.06	97.83	0.17	0.09	0.24
Station B2	0.49	0.99	98.23	0.07	0.05	0.16
Station B3	0.59	0.94	97.83	0.11	0.06	0.48
Station B4	0.51	0.61	98.44	0.20	0.09	0.15
Station B4 ¹	0.54	0.78	98.30	0.16	0.09	0.14
Station B5	0.51	0.87	98.04	0.13	0.11	0.33
Station B5 ¹	0.43	0.65	98.52	0.22	0.12	0.06
Station B6	0.51	0.92	98.39	0.10	0.07	0.00
Station B7	0.46	0.72	98.52	0.15	0.08	0.08
Mean ²	0.51	0.87	98.22	0.14	0.08	0.15

¹ Duplicate analysis of sediment sample

² Mean computed after arc-sine transformation

³ With an average standard deviation [σ (ϕ)] of 0.38, the sediments were close to the very-well-sorted end of the well-sorted range between 0.5 (moderately sorted) and 0.35 (very well sorted)

⁴ Positive skewness

contained a small pebble (Table 4.5). At most stations, the largest fractions consisted mainly of fragments of shell hash and the remains of sand dollars that are randomly distributed on the surface of sediments in many shallow-subtidal coastal areas.

Undetected and Nontoxic Metals

During 2016, most trace-metal concentrations within sediment samples were either not detected or were below levels that are considered potentially toxic to marine organisms (*cf.* mean, ERL, and ERM concentrations in Table 4.6). The only exception was nickel, which is described in more detail below. The presence of mercury and silver was not detected in any of the seven 2016 sediment samples, although they have been detected occasionally in prior monitoring surveys but at concentrations too low to be reliably quantified. This was the case for cadmium in the 2016 samples, which was detected at two of the stations (B5 and B6), but at unquantifiable levels. Thus, the maximum potential concentrations for cadmium, mercury, and silver (0.072, 0.052, and 0.33 mg/Kg) were well below their respective lowest toxic-effects levels (ERLs of 1.2, 0.15, and 1.0 mg/Kg) where biological effects can first become apparent, and far below the median effects levels (ERMs) where biological impacts are expected. Thus, even if cadmium, mercury, and silver were present in Estero Bay sediments in concentrations consistently near quantifiable levels, they would be unlikely to affect marine organisms.

Table 4.6 Metal Concentrations (mg/Kg-dry) within Seafloor Sediment Samples Collected on 25 October 2016

Station	Aluminum	Arsenic	Cadmium	Chromium	Copper	Iron	Lead	Mercury	Nickel	Silver	Zinc
B1	6,900	3.7	<0.063 ¹	69	3.4	10,800	2.3	<0.050	40	<0.32	17
B2	6,300	3.5	<0.063	57	3.8	11,100	2.3	<0.050	49	<0.32	17
B3	6,600	3.5	<0.066	54	3.9	11,100	2.5	<0.052	53	<0.33	19
B4	6,700	3.4	<0.065	53	4.2	11,100	2.5	<0.052	52	<0.33	18
B5	6,800	5.3	≈0.072 ²	56	3.9	10,900	2.4	<0.051	50	<0.33	16
B6	7,000	2.6	≈0.070	60	3.8	11,400	2.3	<0.052	50	<0.33	17
B7	6,100	4.5	<0.067	49	3.5	10,500	2.3	<0.052	47	<0.33	17
Mean ³	6,600	3.7	≈0.067	57	3.8	11,000	2.4	<0.051	49	<0.33	17
ERL ⁴	—	8.2	1.2	81	34.0	—	46.7	0.15	20.9	1.0	150
ERM	—	70.0	9.6	370	270.0	—	218.0	0.71	51.6	3.7	410
SCB ⁵	—	6.4	0.2	30	12.6	—	10.8	—	30	0.2	48

Two of the remaining eight metals, namely aluminum and iron, are not toxic to marine organisms. As described previously, these two metals were added to the monitoring program in 1998 to track natural variations in the concentrations of other trace metals that occur because of inherent differences in

¹ The “less-than” symbol (<) indicates that the substance was not detected at a concentration above the dry-weight-adjusted method detection limit (MDL), which is listed after the “<” symbol.

² The “approximation” symbol (≈) indicates that the reported concentration was too low to be reliably quantified; namely, it was below the practical quantification limit (PQL).

³ After logarithmic transformation

⁴ Guideline not specified

⁵ Concentration that would be considered enriched (99th percentile) relative to background concentrations in the Southern California Bight after normalization by iron (Schiff and Weisberg 1997)

mineralogy. The other six metals (arsenic, chromium, copper, nickel, lead, and zinc) have been regularly detected within Estero Bay sediments throughout the history of the benthic-monitoring program. Although they each have the potential for anthropogenic enrichment, their mean concentrations have remained comparable to ambient levels throughout the central California coastal region. During 2016, as in previous years, five of these six metals had concentrations well below the ERL where biological effects can begin to become apparent (Table 4.6). Only the mean nickel concentration was above the ERL. Each of the six detected metals is described below.

Arsenic

Although present in measurable amounts (≤ 5.3 mg/Kg), the mean arsenic concentration in Estero Bay sediments was less than half of the lowest toxicity level (8.2 mg/Kg) where adverse biological impacts would first begin to appear, and an order of magnitude smaller than the ERM (70 mg/Kg) where biological impacts become probable. For comparison, approximately 75% of the area within the Southern California Bight (SCB) has arsenic concentrations exceeding the mean level measured in Estero Bay during 2016 (Schiff and Gossett 1998). After adjustment for the mean iron concentration, average arsenic levels would have to be 2.7 mg/Kg higher to be considered enriched relative to background concentrations in marine sediments to the south (SCB = 6.4 mg/Kg in Table 4.6).

Pure arsenic is a highly poisonous metallic element used in insecticides, herbicides, solid-state doping agents, and various alloys. However, these anthropogenic sources of arsenic do not constitute a significant component of the wastewater processed by the MBCSD. Instead, most of the arsenic present in the Estero Bay sediments derives from erosion of enriched onshore mineral deposits. Although arsenic occurs naturally in central-coast sediments, mine tailings may contain higher residual levels due to the increased presence of arsenopyrites in the ore, overburden, and soil. Arsenic from mine tailings may be released into surface and ground waters through erosion, and reach Estero Bay through runoff within the adjacent watershed (MRS 2000b).

Chromium

Chromium is also enriched within the sediments of the entire south central-coast region, including within Estero Bay. This enrichment is reflected in the 57-mg/Kg mean chromium concentration collected in samples surrounding the outfall during 2016, which is nearly double the 30-mg/Kg that would be considered impaired for sediments within the SCB (Table 4.6). However, chromium concentrations within all the 2016 sediment samples remained below the lowest toxicity benchmark where biological effects might first be observed. The highest concentration (69 mg/Kg-dry) was measured at reference Station B1, and was still well below the ERL of 81 mg/Kg. More importantly, the maximum concentration was one-fifth of the level where biological effects become probable (ERM=370 mg/Kg). Even at that concentration, the potential for marine effects is probably overstated, given that the incidence of effects in the toxicological studies used to establish the levels for chromium were “*greatly influenced and exaggerated by data from multiple tests conducted in only two field surveys*” (Long et al. 1995).

The ambiguity in chromium’s toxic effects also arises from the fact chromium has several oxidation states, each of which has a vastly different toxicity. The presence of hexavalent and metallic oxidation states of chromium generally arises from anthropogenic sources, particularly industrial processes; however, the relatively non-toxic trivalent form of chromium found within the Estero Bay sediment samples occurs naturally in chromite ore. This is evident from the lack of measurable levels of hexavalent forms in sediment samples collected prior to 1999, when chromium speciation analyses were performed (MRS 1996, 1997c, 1998bi, and 1999b). The elevated chromium levels within Estero Bay sediments originate in naturally occurring chromite minerals that are introduced to the Central Coast’s waters by

fluvial transport of eroded ultrabasic minerals found in the Franciscan formation. The Franciscan formation outcrops along the headlands north of Point Estero and in the Santa Ynez Mountains. Thus, although chromite ore is a widely distributed mineral deposit, it is more prevalent along the south-central coast of California than in most other areas (SAIC 1986).

The presence of higher chromium concentrations within central-coast sediments often leads to misinterpretation by regulators and scientists unfamiliar with the local mineralogy. For example, only around 25% of the SCB has sediments with chromium concentrations higher than those of Estero Bay. Most of this area of higher chromium concentrations lies within Santa Monica Bay, where a high mass-emission of contaminants from publicly owned treatment works occurred prior to 1988. However, elevated chromium levels were also observed within the Santa Barbara Basin in the northern reaches of the SCB. The SBC study speculated that those sediments were impaired as well, when the increased chromium concentrations were actually related to naturally occurring chromite ore within the same geologic formation that extended southward from the south-central California coastal region (Chow and Earl 1978). Their erroneous characterization of an impaired condition in the northern SCB was further invalidated by the incorrect assertion by Schiff and Gossett (1998) that the elevated chromium concentrations were “*highly predictive of adverse biological effects*” in their review of contaminant concentrations within the SCB. However, this assertion is in direct conflict with published findings by the developers of the biological-effects levels, who stated that the chromium effect levels were exaggerated (Long et al. 1995).

While erosion of naturally occurring chromite ore along the central coast explains its overall enrichment within the marine sediments of northern Estero Bay, ordinary erosional processes may not be able to account for the trend of increasing chromium concentrations that was observed in the decade prior to 2000. During that time, increases in chromium, arsenic, copper, and nickel that were unrelated to the MBCSD discharge (MRS 2010a) were documented throughout the Estero Bay region. Although these metals are naturally enriched in onshore mineral deposits, nearshore depositional sediments should have reached equilibrium with onshore sediments that have been eroding within the adjacent watershed throughout geologic time. Instead, the increases in marine sediment concentrations that were observed in the monitoring program during that time suggest that the erosion of these mineral deposits was being accelerated by human activities, such as mining. As described by MRS (2000b, 2001), sediment samples collected in conjunction with a supplementary environmental project indicated that erosion around abandoned chromite mines was the likely cause for increasing metal concentrations within the marine sediments of Estero Bay.

Copper

As with arsenic and chromium, copper commonly occurs within the minerals found along the central coast. However, the 3.8-mg/Kg average copper concentration within marine sediments collected at the seven benthic-monitoring stations during 2016 is well below the 12.6-mg/Kg concentration considered enriched relative to background levels within the SCB sediments. Approximately 90% of the SCB sediments have higher copper concentrations than those measured in Estero Bay. All of the sediment samples collected within Estero Bay during 2016 had copper concentrations that were also far below levels that would begin to affect marine organisms (ERL=34 mg/Kg). In contrast to chromium, the biological effect levels of copper are well defined by toxicological studies. Consequently, biological impacts from the observed copper levels within Estero Bay sediments are highly unlikely given that the highest measured copper concentration (4.2 mg/Kg) was 64-times lower than the ERM level where adverse effects become probable.

Lead and Zinc

Lead and zinc concentrations within Estero Bay sediments are also low compared to ERL levels where adverse biological effects first become noticeable. Additionally, they are low compared to concentrations within the SCB, where approximately 95% of the area has higher concentrations. Most of the elevated lead and zinc concentrations offshore southern California were measured within Santa Monica Bay and on the Palos Verdes Shelf, where large wastewater treatment plants have discharged for many decades. However, most of the metal emissions from these treatment plants occurred prior to the substantial improvements in effluent quality that were realized in 1988. Unfortunately, the historical discharges from these outfalls accumulated within adjacent sediments and have persisted for decades (Stull et al. 1986b). In contrast, the historically low emission of chemicals from the MBCSD treatment plant has had no measurable effect on the high sediment quality within Estero Bay.

Nickel

Although unrelated to wastewater discharge, nickel is the only metal whose concentration within Estero Bay sediments consistently exceeds its ERL level (Table 4.6). In fact, in the three decades of benthic monitoring, none of the 275 samples collected has contained nickel in concentrations less than the ERL of 20.9 mg/Kg. Additionally, twenty-one of the samples have had concentrations exceeding the nickel ERM of 51.6 mg/Kg where biological impacts to marine organisms would normally be expected to occur, including two samples collected during 2016. However, despite measured nickel concentrations that consistently exceed marine-toxicological benchmarks, no apparent deleterious impacts to infaunal organisms living within northern Estero Bay sediments have been observed. There are two reasons for this.

First, nickel exhibits a very weak relationship between the incidence of effects and concentrations in the database used to establish the toxic-effect ranges (Long et al. 1995). Because of this weak toxicological relationship, specification of bulk nickel concentrations that induce adverse reactions in marine biota is highly uncertain. Much of this uncertainty arises from wide variability in nickel bioavailability. When nickel is in a dissolved form, or adheres to the surfaces of fine-grained sediments, it is much more likely to affect organisms that ingest or come into contact with the sediments. Conversely, nickel that is bound into the mineralogy of larger sand grains, as is the case within MBCSD benthic monitoring area, has little influence on marine organisms.

The second reason for the lack of impacts from elevated nickel concentrations stems from the fact that nickel occurs naturally in the mineralogy of the southern portion of the central California coast (Steinhauer et al. 1994). Because nickel and chromium often co-occur in mineral deposits, it is likely that the increased nickel concentrations found offshore also derive from the Franciscan geologic formation that outcrops at Point Estero. Thus, much of the measured nickel is bound into the mineral matrix of the sand grains themselves, where marine fauna cannot readily metabolize it. Additionally, if there is a slightly higher dissolved-phase nickel concentration within the pore waters of Estero Bay, it has probably persisted for epochs, and benthic organisms in the region may have adapted accordingly.

Nickel and chromium are the only metals whose measured bulk concentrations in Estero Bay sediments consistently exhibit “*anthropogenic enrichment*” relative to background concentrations in the SCB (Schiff and Weisberg 1997). However, this designation is incorrect because much of the regional enrichment along the central California coast arises from natural erosional processes, rather than from accelerated erosion of contaminated mine tailings. In fact, the influence of regional mineralogy is also plainly evident in the SCB database where “*background*” nickel concentrations steadily increase from south to north. As with chromium, Schiff and Gossett (1998) did not account for this natural variability in ore deposits and erroneously ascribed observed nickel gradients within the northern SCB to anthropogenic contamination.

Organic Loading

Organic-loading parameters including moisture content, mud fraction, BOD, and O&G measured in the sediment samples collected during 2016 were comparable to levels measured in most benthic surveys (Table 4.7). Although TVS concentrations measured within Estero Bay sediments from 2014 through 2016 were perceptibly higher than in previous years, this may be an artifact of the high numbers of large sand dollars that were present during that time. In contrast to trace-metal concentrations, it is difficult to gauge the significance of the organic constituent concentrations measured during 2016, and in prior years, because there are no well-established biological effects levels for any of these parameters.

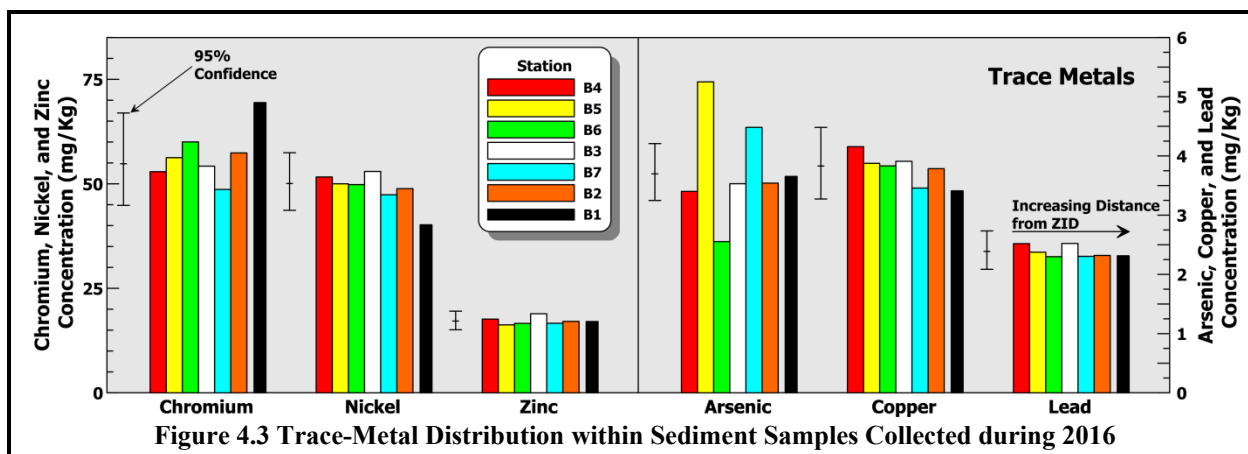
4.2.2 Spatial Distribution Unrelated to Outfall Proximity

During 2016, as in previous years, the spatial distribution of sediment chemicals did not exhibit a pattern indicative of a benthic environment impaired by the MBCSD wastewater discharge. Specifically, there was no evidence that chemical concentrations within sediment samples collected near the diffuser structure were elevated relative to those of distant sites. A gradient of steadily increasing contaminant concentrations with increasing outfall proximity would provide *prima facie* evidence of discharge-related impacts to the marine environment. This pattern of impairment has been observed around larger outfalls servicing more industrialized locales (Phillips and Hershelman 1996, Stull et al. 1986b, Stull 1995, Diener et al. 1995). The lack of contaminant loading around the MBCSD outfall was confirmed through application of rigorous statistical hypothesis tests, and by qualitative visual comparisons of concentrations ranked by distance from the diffuser structure.

The bar graphs in Figure 4.3 show trace-metal concentrations ranked by distance from the diffuser. Potential outfall-related impacts would appear as bar heights steadily decreasing from left to right in many, if not all of the trace metals. However, concentrations measured at the geographically distant

Table 4.7 Non-metal Concentrations (mg/Kg-dry)¹ within Seafloor Sediment Samples Collected on 25 October 2016

Station	Moisture (%)	BOD	Oil & Grease	TKN	Volatile Solids (%)
B1	17.9	50	— ²	183	1.81
B2	18.1	45	<17	195	2.25
B3	20.7	50	<18	202	2.46
B4	20.6	60	<18	214	2.51
B5	20.0	51	<18	213	2.33
B6	21.7	60	<18	192	2.27
B7	21.9	55	<18	166	2.15
Mean ³	20.1	53	<18	194	2.25



¹ Unless otherwise indicated as “%”

² Sample not analyzed for O&G

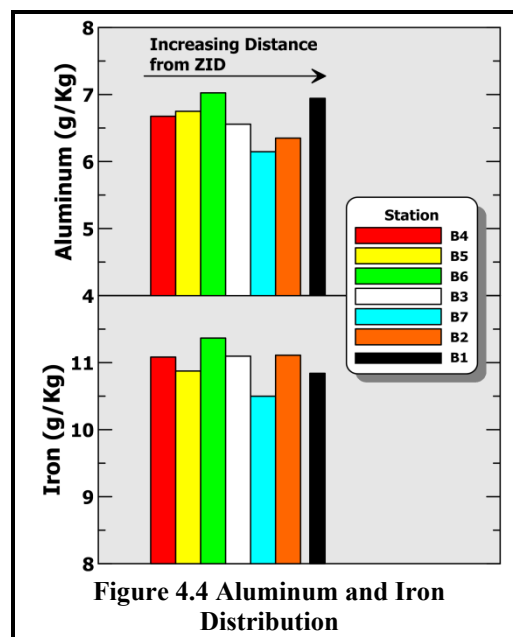
³ Arc-sine transformation for % and logarithmic transformation for mg/Kg

Station B1, shown by the thinner, separate black bars in Figure 4.3, can be disregarded. As described in the last paragraph of Section 4.1.1 on Page 4-3, sediment samples collected at Station B1 are considered irrelevant for compliance purposes because its benthic environment has been found to be materially different from that of the permit-required survey area surrounding the outfall.

Visual inspection of the concentration sequences at the remaining sites shown in Figure 4.3 do not readily exhibit an enrichment gradient potentially related to outfall proximity, except for perhaps copper. Although copper concentrations generally appear to decrease with increasing distance from the diffuser structure (left to right in the Figure), the gradient was so slight that the actual spatial differences among all the stations were encompassed within the uncertainty in concentrations measured at individual stations. This uncertainty is quantified by the 95% confidence intervals shown to the left of each bar graph.¹ They indicate that perceived differences in concentrations measured among stations during 2016 were smaller than the inherent sampling error, and by definition, have a low degree of confidence.

When all the measured metal concentrations lie within the 95% confidence interval associated with that metal, then any observed differences in concentrations among stations can be ascribed to inherent sampling variability, rather than to actual spatial changes in chemical concentrations within the sediments. Figure 4.3 shows that, during 2016, all of the individual metal concentrations measured within the survey area² were encompassed within their respective confidence interval, except for arsenic. In contrast to the other metals, and in contrast to prior surveys, arsenic concentrations during 2016 were unusually variable with markedly high concentrations at Stations B5 (yellow bar) and B7 (blue), and a strikingly low concentration at the intervening Station B6 (green). These three stations are the only ones located on south side of the diffuser structure and were the first to be sampled. Although there is no ready explanation for the unusual scatter in the concentrations measured at these three stations, it is clear that they were not related to outfall proximity.

Inherent sampling variability aside, some of the scatter in the raw metal concentrations may have resulted from natural processes unrelated to the outfall discharge. As described in the last paragraph of the *Trace Metals* subsection within Section 4.1.4 on Page 4-8, the influence of natural processes on the distribution of sediment-metal concentrations can be reflected in the aluminum and iron distributions (Figure 4.4). Human activities have little influence on aluminum and iron concentrations within seafloor sediments and normalization by these concentrations is a common practice in impact-assessment studies because it can reveal anthropogenic patterns that are otherwise masked by variability in background trace-metal concentrations. Similarly, they can eliminate spatial

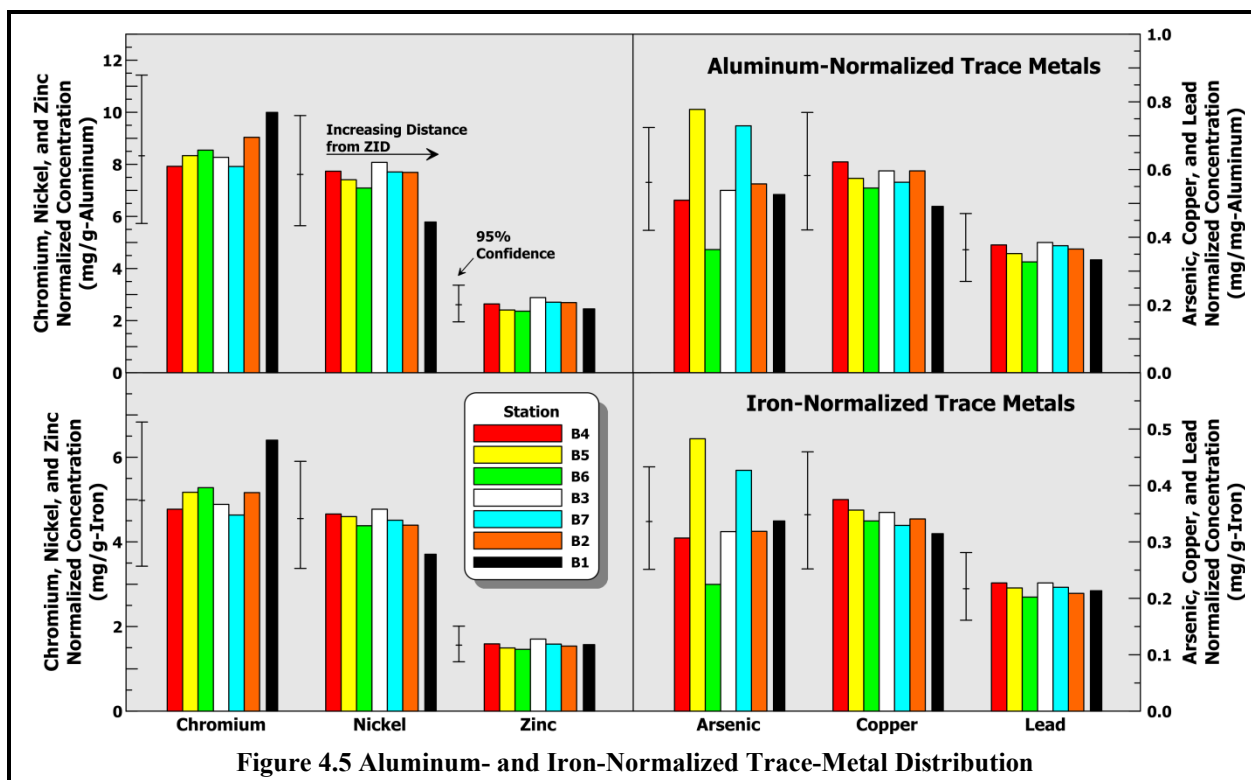


¹ The 95% confidence intervals reflect the inherent sampling uncertainty at any given station and were determined from the variability in concentrations found among three individual replicate samples collected and separately analyzed at each station during 1993 and 1994.

² As described previously, Station B1 is located far beyond the boundary of the survey area in a location where the benthic environment is measurably different. Not surprisingly then, metal concentrations measured in sediment samples collected at Station B1 regularly depart significantly from the average of the concentrations within the survey area. Depending on the metal, the Station B1 concentration can be above (chromium) or below (nickel) the survey-area average.

differences that may appear to be related to a discharge impact, when they are in fact due to natural variability.

Although the aluminum and iron spatial patterns shown in Figure 4.4 do not match any of the distributions in Figure 4.3 particularly well, normalization (Figure 4.5) appears to temper the variability in some of the metals relative to the confidence intervals. Additionally, the subtle gradient of decreasing copper concentrations with distance from the outfall seen in the raw data (second to the last bar chart in Figure 4.4) is eliminated in aluminum-normalized concentrations (upper right frame of Figure 4.5). Normalization did not, however, reduce or eliminate the unusual scatter in arsenic concentrations, indicating that the excursions were not the result of inherent variability in overall ambient trace-metal concentrations at the three southernmost stations.



Although spatially ranked bar charts provide a useful visualization of potential gradients, statistical hypothesis tests provide a more rigorous, quantitative method for assessing the significance of potential spatial gradients such as the subtle gradient in raw copper concentrations. Specifically, these tests establish whether the MBCSD discharge could be causing an increase in metal concentrations close to the outfall compared to the more distant stations. Two types of spatial tests were performed on the data collected during 2016 (Table 4.8). First, a test for the presence of a spatial gradient was performed by fitting a line to the concentration versus distance data. If the concentrations consistently decrease with distance, then the negative slope of the line will be statistically significant and possibly indicative of outfall-related impacts.

The second statistical test evaluated whether mean concentrations at the three close-in stations (B4, B5, and B6) where samples were collected within 35 m of the discharge differed significantly from the average of the three distant stations (Stations B2, B3, and B7), that are located more than 65 m away. In contrast to the first type of test for spatial gradients, the second hypothesis test did not evaluate whether

Table 4.8 Tests for Higher Contaminant Concentrations near the Diffuser

Parameter	Linear Trend		Difference in Means (mg/Kg)				Iron-normalized ¹ <i>p</i>	Aluminum-normalized <i>p</i>
	Slope	<i>p</i> ²	Near ³	Distant ⁴	<i>t</i> ⁵	<i>p</i> ²		
Aluminum (g/Kg)	-0.041	0.09	6.81	6.35	2.92	0.03	0.02	NA
Arsenic	0.020	[0.90]	3.57	3.83	-0.31	[0.39]	[0.43]	[0.35]
Chromium	-0.019	0.67	56.3	53.3	0.91	0.22	0.23	[0.37]
Copper	-0.054	0.08	3.95	3.71	1.40	0.13	0.16	[0.44]
Iron (g/Kg)	-0.011	0.51	11.1	10.9	0.83	0.23	NA	[0.02]
Lead	-0.024	0.36	2.40	2.38	0.17	0.44	[0.4]	[0.14]
Nickel	-0.024	0.30	50.5	49.7	0.46	0.35	[0.46]	[0.08]
Zinc	0.005	[0.88]	16.8	17.5	-0.89	[0.22]	[0.12]	[0.04]
BOD	-0.083	0.19	57.1	50.1	1.67	0.10		
TKN	-0.079	0.10	206	187	1.36	0.13		
TVS (%)	-0.016	0.16	2.37	2.28	0.73	0.26		
Mud (%)	0.030	[0.04]	1.21	1.48	-8.68	[0.002]		

the concentrations steadily changed along a gradient; instead, it assessed whether the average concentration in the group of stations close to the outfall was significantly higher than that of distant stations, which would again be indicative of outfall-related impacts.

The “*p*-value” determines the significance of any perceived spatial gradient by testing it against a null hypothesis of no gradient. It quantifies the probability that a perceived spatial gradient could have occurred by chance. When $p < 0.05$, there is less than a 5% risk that the observed spatial gradient was happenstance due to random fluctuations. This threshold for designating potentially significant spatial gradients is consistent with the 95% confidence levels promulgated by the COP and used throughout this report. However, it is important to keep in mind that during any given year, a sediment constituent may exhibit an anomalous concentration at one or more stations that results in a spatial gradient that appears to be related to outfall proximity, for example, when that gradient is determined to be statistically “*significant*” at the 95% confidence level. Nonetheless, by the very definition of the 95% confidence level, one in twenty tests would be expected to reveal a “*significant*” gradient when, in fact, none exists.

Over the years, many constituents have been tested for spatial gradients, and occasionally the analysis identifies one that appears to be statistically significant at some particular time. Upon further analysis, however, the gradient has always been found to be unrelated to the discharge. Additionally, approximately half of the time, the statistically “*significant*” gradients reflect decreasing concentrations

¹ Results of hypothesis tests performed on trace-metal concentrations normalized by the iron and aluminum concentration to reduce the potential influence of inherent variability in background metal concentrations.

² *p*-value or probability that a linear gradient is not significant or that a higher nearfield mean concentration is not significant (null hypotheses). In this study, *p*-values less than 0.05 (95% confidence) indicate that there is a linear gradient or that nearfield means are significantly higher, and the null hypothesis can be rejected (shown in bold font). Note, however, that a “*one-sided*” *t*-test was performed, in which only higher, rather than lower, nearfield concentrations are meaningful. Consequently, *p*-values for negative *t*-values are not pertinent and are indicated with brackets.

³ Mean concentration computed among Stations B4, B5, and B6, which lie within 35 m of a diffuser port (the largest closest-approach of chemistry grabs in Figure 4.2). For percent concentrations, an arc-sine transform was applied to the concentrations prior to hypothesis testing. For other units, a logarithmic transform was applied.

⁴ Mean concentration computed among Stations B2, B3, and B7, which lie more than 65 m from a diffuser port.

⁵ *t*-value computed from the difference in near and far mean concentrations relative to measurement variability. A negative *t*-value indicates that the computed mean concentration was lower at stations near the diffuser, opposite of that expected in an environment impacted by effluent discharge.

with increasing proximity to the outfall, which is opposite the spatial gradient expected from impacts associated with effluent discharge.

Within the 2016 database, none of the sediment metals exhibited a statistically significant linear gradient (first two data columns of Table 4.8). Among the metals, the lowest p -value of 0.08 (second data column) was determined for the aforementioned visual impression of decreasing copper concentrations with increasing distance from the outfall. However, the gradient, as measured by a linear slope of -0.054 (first data column), was not large enough or consistent enough to be statistically significant, namely, with a p -value less than or equal to 0.05. Arsenic and zinc exhibited a positive slope, where concentrations increase with distance from the outfall. Because this gradient is opposite of that expected from a discharge-related impact, their p -values are enclosed in brackets, indicating the test is irrelevant to the compliance assessment.

The results for the other type of test, for significant differences in near and distant mean concentrations, are shown in the remaining columns of Table 4.8. In this case, that average raw aluminum concentration at the three close-in stations (6.18 mg/Kg) was the only metal found to be significantly higher than that of the three distant stations (6.35 mg/Kg). Because the associated p -value (0.03) was less than the significance threshold (0.05), it is highlighted in bold in the Table. The significance of the difference is not surprising given that all three of the close-in stations (red, yellow, and green bars in the top frame of Figure 4.4) had higher aluminum concentrations than the three distant stations (white, blue, and orange bars). Regardless of its significance, the aluminum distribution results from natural processes and therefore, is not subject to direct compliance evaluation.

The same spatial tests can be applied to the distributions of the four organic loading parameters (Figure 4.6). As with the metals, the scatter in their concentrations among the stations within the survey area are encompassed within the sampling uncertainty associated with measuring a concentration at any given station (95% confidence limits to the left of each bar graph). Additionally, there are no obvious spatial differences related to outfall proximity in the rank-ordered sequence of bars, except for perhaps the lower mud concentrations found at stations nearest the outfall.

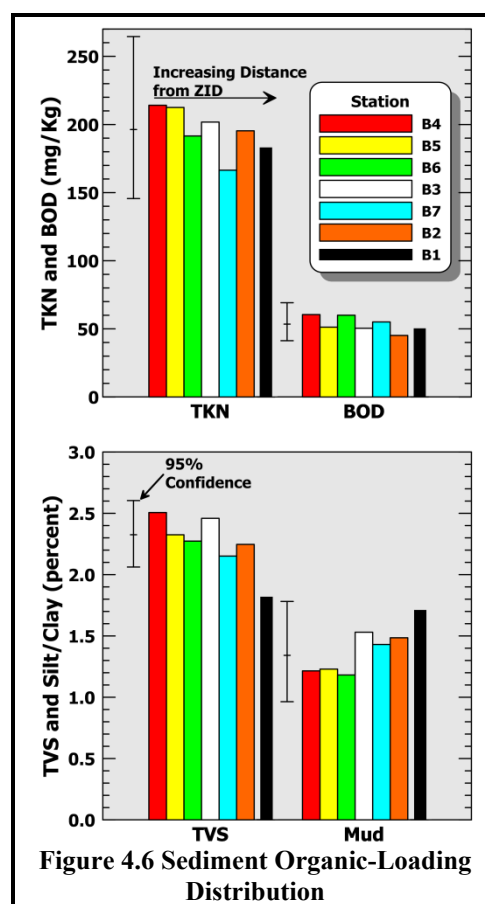


Figure 4.6 Sediment Organic-Loading Distribution

Accordingly, none of the spatial tests found statistically significant linear gradients or differences in average concentrations at near and distant stations for BOD, TKN, and TVS ($p \geq 0.10$ in Table 4.8). However, both types of tests identified the decreased mud fraction at close-in stations as significant. However, because there was no steady mud increase with increasing distance, the linear gradient test was only marginally significant ($p=0.04$). In contrast, because all the mud fractions measured in the nearfield samples were very close to one another, as were the distant sediment samples, the difference in mean concentrations was highly significant ($p=0.002$).

The significance of this mud distribution in the 2016 dataset, and the fact that noticeably lower mud fractions near the outfall have been seen in many prior surveys, suggests that mud content actually declines in relation to outfall proximity, at least in some sampling surveys. However, reduced mud content near the outfall is opposite of the spatial distribution expected from an accumulation of wastewater particulates within sediments adjacent to the discharge. As such, it is not related to the quality of the effluent, its impact on the receiving waters, or the assimilative capacity of the marine environment as a whole.

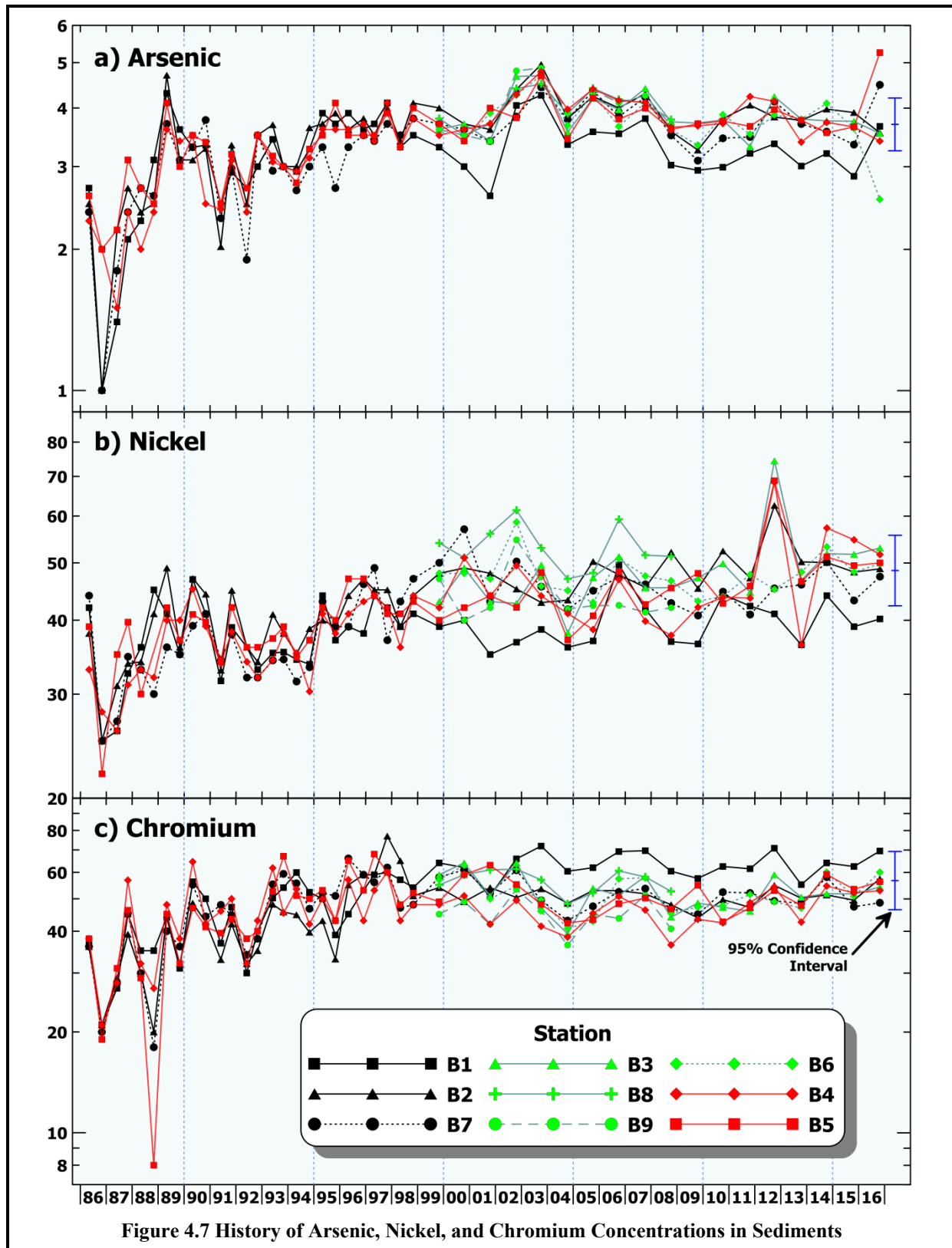
As with the other gradients that are immaterial to compliance assessments associated with the NPDES discharge permit, the *p*-values determined from the mud-fraction tests are enclosed in brackets in (Table 4.8). Insight into the cause of the lower mud fractions near the outfall may have been provided by past surveys where benthic samples were accidentally collected very close to the outfall. Reduced mud fractions resulting from scour around the outfall structure were found to cause a sharp decrease in mud fraction in those samples. Thus, the physical presence of the outfall structure on the seafloor is capable of altering the grain-size distribution, although this effect has not been known to extend very far from most seafloor structures in general.

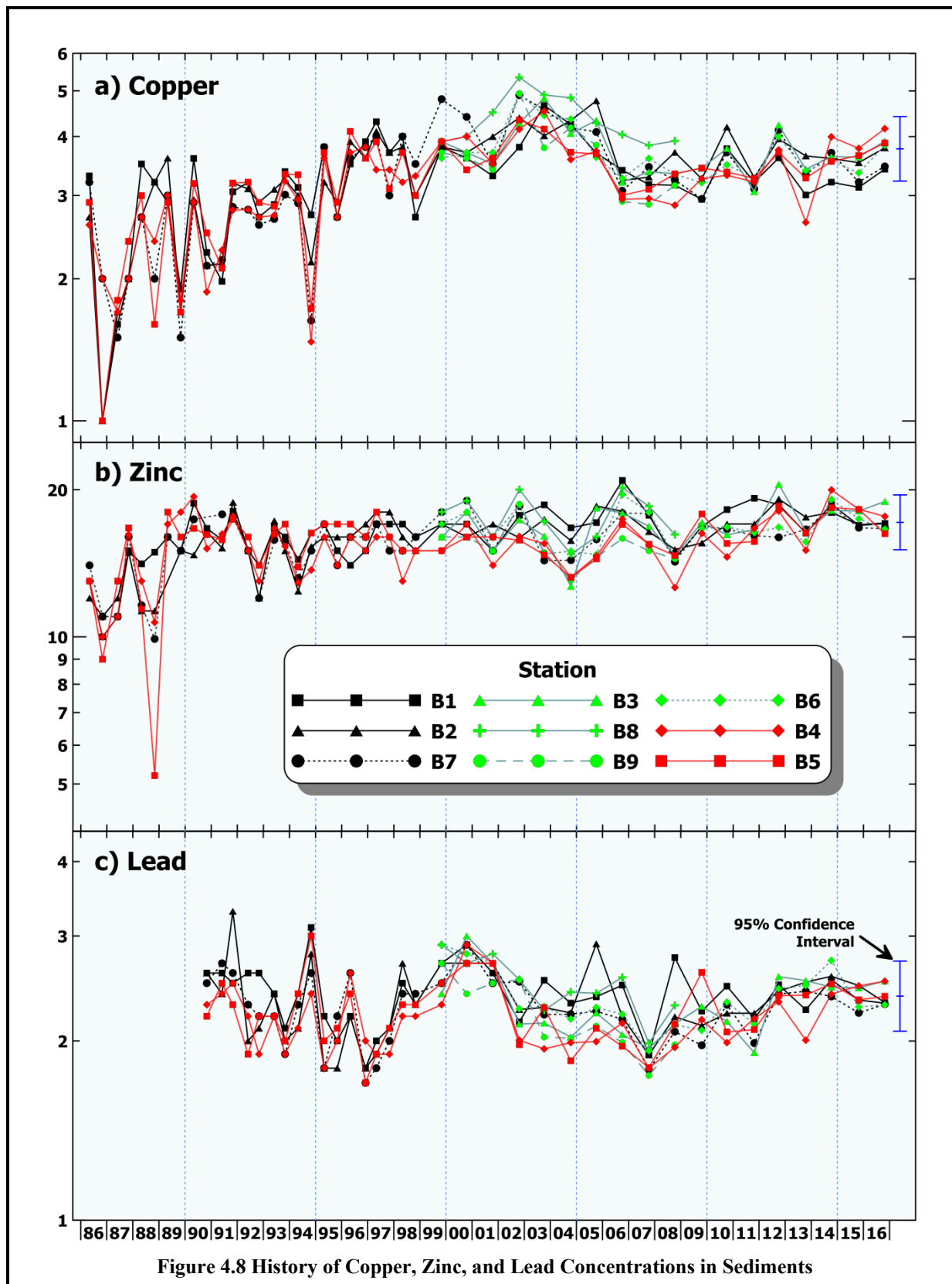
4.2.3 No Buildup over Time Related to Wastewater Discharge

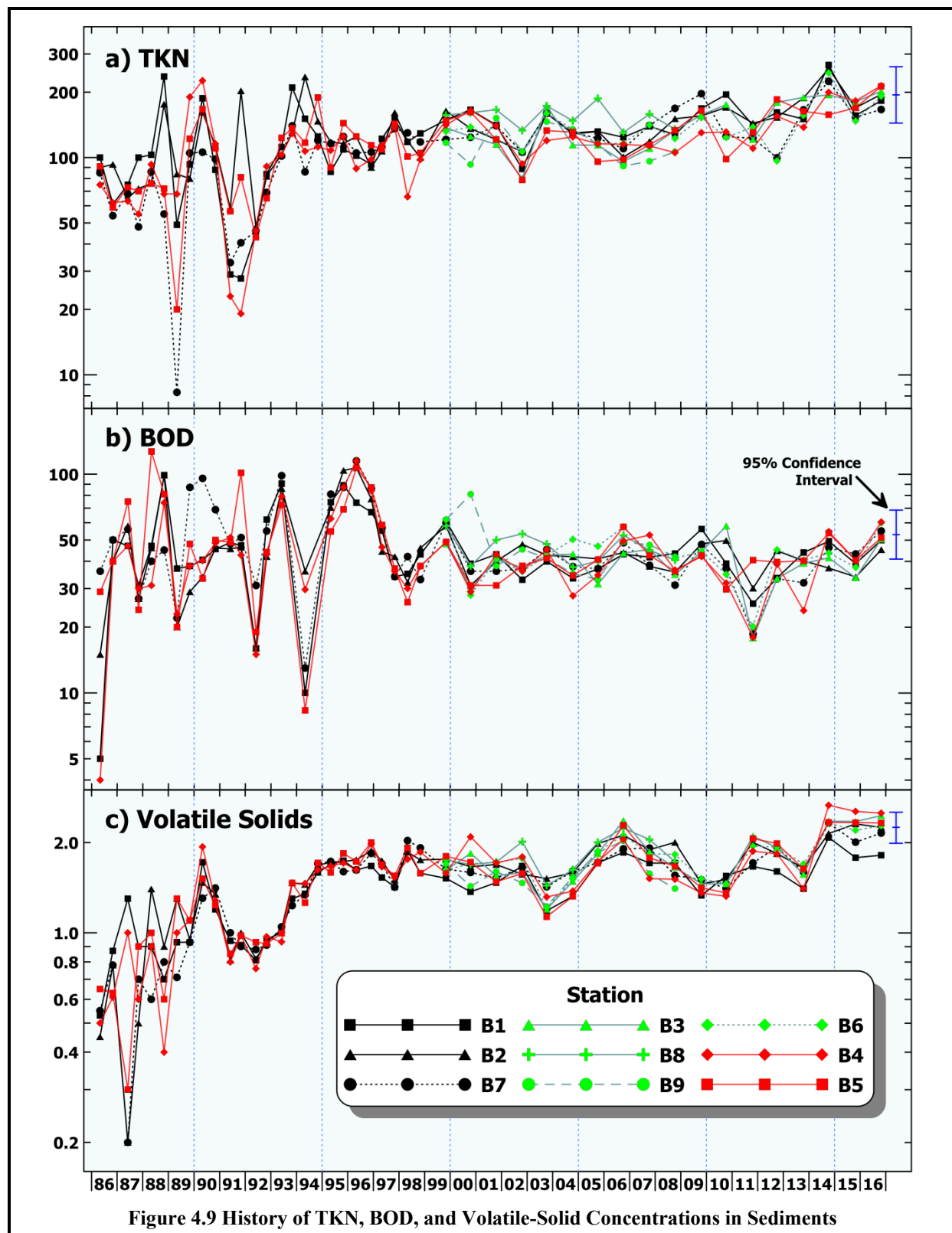
The previous section examined the spatial distribution of sediment constituents at a single point in time, namely, during the October 2016 survey. However, the characteristics of ambient sediments at individual stations can be inherently different because of slight differences in their water depth, or because of differences in the depositional characteristics of their geographic location. These factors complicate the interpretation of any perceived spatial gradients with respect to effluent discharge. Examining changes in spatial distributions over time, however, significantly improves the power to detect potential discharge-related impacts. Specifically, if there were any measurable marine impacts from wastewater discharge, they would be revealed by a long-term buildup of contaminants in sediments surrounding the outfall, as compared to distant stations. Consequently, examining spatial gradients in contaminant concentrations over time can reveal discharge-related trends that may be masked in a spatial analysis applied to a single survey.

However, perceived temporal trends in contaminant concentrations must be interpreted carefully because natural and anthropogenic processes unrelated to the effluent discharge can also cause large and widespread changes in chemical concentrations within sediments. These regional influences are apparent in the time series of sediment chemical concentrations shown in Figure 4.7, Figure 4.8, and Figure 4.9. Specifically, the large-amplitude fluctuations that occur over periods of a few years reflect widespread seasonal and interannual changes in the sedimentary environment. Similarly, general increase in the concentrations of five sediment constituents that are apparent in the decade-and-a-half prior to 2000 are in response to even longer-term influences within northern Estero Bay (Figure 4.7abc, Figure 4.8a, and Figure 4.9c).

Nevertheless, these large interannual fluctuations and long-term increases tend to occur in unison at all stations, so the spatial differences among samples collected at a given time (i.e., within a given survey) are comparatively small. In other words, the fluctuations in concentrations generally track one another through time. This indicates that changes in sediment concentrations that arise from regional influences throughout Estero Bay are far larger than those induced by any localized effect. During a given survey, spatial differences in sediment concentrations tend to remain within the 95% confidence interval (shown to the right of each time series in the figures) that quantifies the uncertainty in individual measurements.







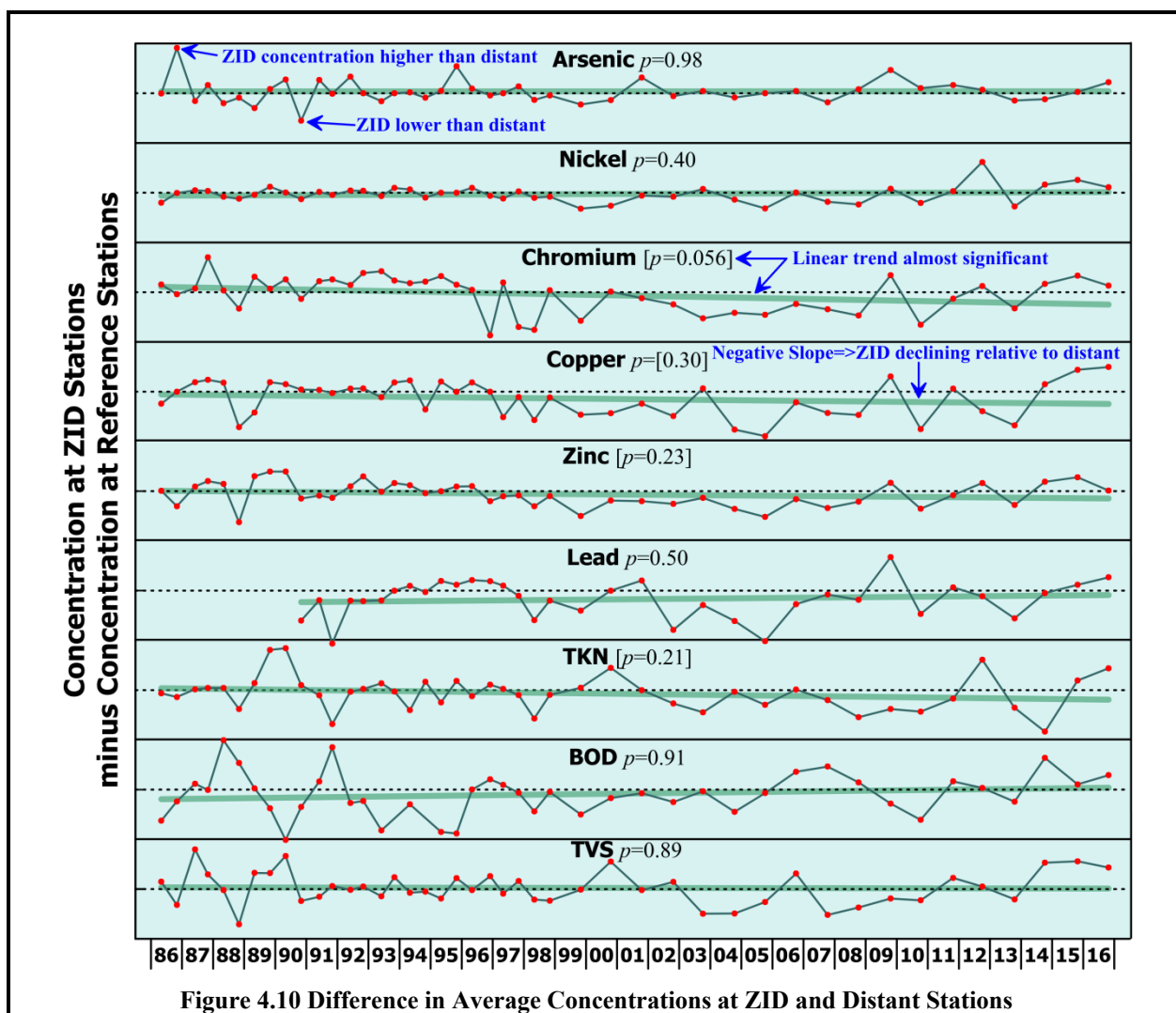
As described in the previous section, these confidence bounds indicate that most spatial differences within a given survey are not statistically significant, while interannual fluctuations and long-term temporal trends are large enough to be reliably quantified in the presence of sampling uncertainty. Moreover, these large temporal changes can mask subtle spatiotemporal trends in sediment chemical composition that may be related to the MBCSD discharge. However, the parallelism techniques applied in this section can reveal relevant spatiotemporal relationships that would otherwise be difficult to discern in the presence of these large regional excursions in sediment chemistry.

The pre-2000 long-term increase in the sediment concentrations of a number of metals, predominately arsenic, chromium, nickel, and copper, was the subject of intensive investigation in prior annual reports (MBCSD 1999b, 2000a, 2001 – 2010) and in an independent, separate field study (MRS 2000b). Those analyses unequivocally demonstrate that the observed increase could not be caused by the MBCSD discharge because: 1) the accumulation occurred at all stations, including stations distant from the outfall, 2) the MBCSD metals emission was far too small to account for the widespread increase, and 3) the discharge remained unchanged while the temporal trend in average concentrations leveled out after 2000. Instead, the steady increase in sediment metal concentrations prior to 2000 was ascribed to accelerated erosion of contaminated mine tailings at abandoned chromite mines within the adjacent watershed. After 2000, sediment concentrations appear to have reached equilibrium with the metal loading from the watershed's runoff into Estero Bay.

Discerning subtle localized impacts within the backdrop of large interannual fluctuations and long-term temporal trends is aided by a special statistical technique that tests for the presence of parallel time histories. Parallelism tests overcome limitations associated with purely spatial or purely temporal approaches to impact evaluation (Coats et al. 1999, Skalski et al. 2001). This technique assumes that the large temporal excursions in concentrations occur in unison at all stations and represent a simultaneous response to regional influences unrelated to the MBCSD discharge. Under those circumstances, discharge-related spatiotemporal changes would appear as a gradual separation of the colors in Figure 4.7, Figure 4.8, and Figure 4.9, as contaminants preferentially accumulate near the outfall over time. Concentrations at nearfield stations (shown in red) would slowly increase relative to concentrations at mid-field (green) and distant reference stations (black). However, because of the large coherent fluctuations that occur over time, subtle spatiotemporal changes may be present but not visually apparent in the time series.

Analysis of the difference between average concentrations at stations located near the outfall, and those that are distant from the outfall, effectively removes the temporal oscillations that occur on a seasonal and interannual basis. Specifically, if the time series of average concentrations near the outfall parallel those of distant stations, then there is little evidence of a discharge-related accumulation. A parallelism test is also insensitive to inherent differences in the sedimentary environment at the two groups of stations. Distant stations may have consistently higher (or lower) average concentrations because of natural spatial differences in the benthic environment, but because the parallelism test only deals with trends in the difference between the two groups of stations, rather than their absolute values, such natural spatial variations are appropriately eliminated from the compliance evaluation.

Time series of differences in sediment concentrations at ZID and distant stations are presented in Figure 4.10. If contaminants discharged by the MBCSD discharge were accumulating in sediments near the outfall, then concentrations at stations nearest the outfall (B4 and B5) would slowly increase relative to distant stations (B2 and B7). These increases would appear as positive slopes in the difference in averages shown in Figure 4.10. A regression test for a statistically significant trend (slope) in the difference is equivalent to a test of the null hypothesis that the contaminant concentrations at impacted and non-impacted sites remain constant (parallel) over time.



Five compounds exhibited slightly positive slopes, suggesting the possibility of a buildup of wastewater contaminants close to the discharge compared to stations that are more distant. Because this trend may be of compliance interest, their p -values were not enclosed in brackets in the difference series of Figure 4.10. Similar to the other statistical tests in this report, the p -values measure confidence in the slope of the line, by comparing the slope magnitude with the variability resulting from the presence of scatter among individual data points. When the slope is small or the scatter is large, the spatiotemporal trend cannot be considered statistically significant. This was the case for all five of the compounds with positive slopes because the tests indicated that all of these spatiotemporal trends were far too small to be reliably determined in the presence of year-to-year fluctuations ($p \geq 0.4$).

The remaining four chemicals exhibited negative slopes in Figure 4.10, indicating that ZID concentrations were possibly decreasing over time relative to the reference stations. Because this trend is opposite of that expected from a buildup of contaminants near the discharge, the parallelism hypothesis test is irrelevant for a compliance evaluation, so the associated p -values for the parallelism test are listed with brackets surrounding them in Figure 4.10. Additionally, the spatiotemporal trends in three of the compounds were very poorly resolved ($p \geq 0.21$), as was the case for the five compounds with positive slopes.

The slope of one chemical, however, was nearly large enough to be resolved (third plot from the top in Figure 4.10). Specifically, the hypothesis of a long-term trend in relative chromium concentrations was only marginally rejected at the 95% confidence level ($p=0.056$). In fact, prior to the last data point that was added in 2016, which showed a slightly higher concentration at the ZID stations, chromium exhibited a strong case for rejecting parallelism ($p=0.031$), and accepting the alternative hypothesis that chromium concentrations are actually decreasing near the outfall compared to distant stations. Before the three positive differences were added in the last three years, the spatiotemporal trend through 2013 was strikingly well established ($p=0.003$).

This negative chromium slope could be indicative of large-scale changes within the sediments of Estero Bay that may warrant attention even though they are unrelated to the MBCSD discharge. The parallelism tests in Figure 4.10 included four stations within the survey area immediately surrounding the outfall¹ and thus, only represented a small portion of northern Estero Bay region. As a result, the separation of chromium concentrations among these stations was too subtle to discern from an inspection of the time series shown in Figure 4.7c. However, chromium concentrations at the northernmost Station B1, shown by the black trace with square data points, exhibit an obvious temporal separation that is consistent with the parallelism tests. In the first half of the record, chromium concentrations were comparable to those of other monitoring stations. In the latter half of the record, however, chromium concentrations at Station B1 diverge from rest of the group. Specifically, in each of the last fourteen years, concentrations of chromium have been higher at Station B1 than at any of the other stations. This overall spatiotemporal trend in chromium concentrations may be a manifestation of ongoing erosion of mine tailings that caused the widespread increase in some metal concentrations prior to 2000. The slightly higher chromium concentrations at Station B1 may reflect more localized effects due to its proximity to the mouth of Toro creek, whose watershed encompasses a relict chromite mine.

4.3 BENTHIC INFAUNAL COMMUNITY

The infauna living within the surficial sediments of northern Estero Bay provide additional insights into the wellbeing of its seafloor environment. They are also particularly effective for compliance determinations that assess the presence of a BIP surrounding the outfall. They act as sentinels that are responsive to possible impacts caused by wastewater discharge that may be too minor to be revealed in the analysis of chemical constituents alone. Infauna are the best biological indicators of marine pollution because they have limited mobility and cannot easily escape exposure to contaminants in their immediate environment. Other organisms, such as marine mammals and finfish, range widely, making it impossible to determine where contaminant exposure may have occurred.

The diffuser structure lies on the seafloor, where particulate contaminants discharged into marine waters ultimately settle and accumulate. Because infauna reside within seafloor sediments, they are close to the potential source of pollution. Additionally, individual infaunal species vary in their sensitivity to pollutant stressors. As such, changes in relative abundance among species can signal the presence of subtle alterations in their environment. Infauna are also important organisms to monitor because of their low trophic level within the marine food web. They provide food for more-mobile epifaunal and pelagic marine organisms, such as crabs, finfish, and marine mammals.

¹ As with the other statistical tests performed on sediment constituents in this chapter, Station B1, which lies well north of the survey area, was excluded because its benthic environment is not representative of the area around the discharge. The parallelism tests in Figure 4.10 compared average concentrations at ZID Stations B4 and B5, shown in red, with midfield Stations B2 and B7, shown by the black traces with round and triangular data points.

In recognition of infauna's role as an early detector of marine pollution, the MBCSD discharge permit requires regular monitoring of the overall health of the benthic community within Estero Bay. As such, infaunal monitoring addresses the COP requirement that the discharge not degrade indigenous marine biota (see Objective E.1 in Table 3.1 on Page 3-3). The benthic monitoring has now amassed more than three decades of infaunal data. During that time, 262,000 specimens representing 392 individual taxa have been collected, identified, and enumerated. Throughout the monitoring program, there has never been an indication of deleterious discharge-related impacts to the benthic community. Instead, the data have revealed a consistently healthy indigenous infaunal community with uniformly high diversity beyond the ZID, and no evidence of degradation in the infaunal assemblages related to outfall proximity.

These observations hold true despite widespread variations in the abundance of individual organisms. Over the years, the dominance of individual taxa have waxed and waned in response to natural changes in the environment. Additionally, for many taxa, distinct differences in abundance are evident between the winter and summer sampling seasons. Other species come and go in response to powerful long-term variations in the oceanographic environment. One such variation is the El Niño – Southern Oscillation, whose impact has been evident in the benthic environment of northern Estero Bay on at least four occasions during the last three decades of MBCSD monitoring (NOAA 2010). The first event was in 1988, and the second began at the end of 1997 and extended into 1998. Its influence on benthic organisms was still evident during 1999. Another minor event may have occurred in 1991; however, the event with the largest influence on faunal populations around the outfall occurred in 2009 when the faunal density was 100-times higher than historical average due to a huge explosion in the sand-dollar population.

The effects of this initial population boom continued to be apparent through 2016 as the original 2009 cohort matured and grew in size. As described in the 2015 annual report (MRS 2016a), that sand dollar cohort began to profoundly impact the entire infaunal community when numerous mature sand dollars (*Dendraster excentricus*) dramatically altered the benthic environment and displaced a large fraction of the sediment that is normally collected within each grab. As described in this report, the continued presence of numerous large sand dollars during 2016 continued to affect the infaunal community profoundly. However, as described below, the initial onset of sand-dollar senescence began to affect the resident infauna in subtly different ways.

Over 4,126 individual infaunal organisms were enumerated in the 35 sediment samples that were collected during the benthic survey conducted during October 2016. A compendium of the taxonomic identifications and enumerations that were performed on the year-2016 sediment samples is provided in Tables B.1 through B.8 in Appendix B at the back of this report. Tables B.9 and B.10 list community indices that were computed from infaunal data, as required by the NPDES monitoring and reporting program. This section of the annual report combines these data with biological data collected over the prior three decades to determine whether there have been discharge-related impacts to the benthic biota.

Three quantitative analyses were conducted to evaluate whether there have been deleterious impacts to benthic organisms as a result of the discharge. First, the organisms that inhabit the sediments around the outfall were evaluated to determine whether they indicative of an indigenous community. Throughout most of the monitoring program, indigenous infauna were dominated by filter-feeding organisms that thrive only in clean sediments, but during 2015, the entire benthic community underwent a major transformation to one more typical of dense sand-dollar beds. Second, infaunal data collected during 2016 were assessed for evidence of spatial gradients indicative of biotic degradation attributable to effluent discharge. Third, the variations in time histories of community parameters were examined for long-term spatiotemporal trends indicative of an increasingly degraded benthic environment around the outfall. The

historical record shows that infauna at individual stations tend to respond in unison to large interannual and seasonal influences on the infaunal community. As a result, infaunal differences among stations during a given survey tend to be negligible compared to naturally occurring temporal variations. To date, the indigenous benthic community remains healthy, with no perceptible influence from ambient contaminant levels, including the previously discussed long-term temporal trends in some trace-metal concentrations that occurred throughout Estero Bay prior to 2000.

4.3.1 Excluded Organisms

In the past, fluctuations in the population of two organisms, *Diopatra ornata* and *D. exentricus*, have had a profound effect on the MBCSD infaunal database. However, these organisms and their attached taxa are not particularly diagnostic of changes in sediment quality, and their presence frequently confounds impact assessments. Consequently, their populations were removed from the database prior to the impact assessments described in this section.

Tube Worms

Since 2000, the parchment tubes of the Ornate Tubeworm (*D. ornata*) have routinely been separated in the field prior to sieving of the infauna. The “Ornate” portion of the tubeworm’s common name refers to the elaborate layering of pieces of shells, algae, sticks, and other debris on the outside of their large parchment tubes (Figure 4.11). Large numbers of nestling epibionts reside on the microhabitats created by these tube casings, including small crabs and other crustaceans, particularly Gammarid amphipods. Inclusion of these epibionts in the infaunal database can dramatically skew the enumeration of marine infaunal organisms in samples where they are randomly present. Removal of the *D. ornata* parchment tubes in the field, prior to sieving, limits sample contamination by the epibionts associated with these tubes. The tubeworms themselves, however, are included in the enumerations because they tend to congregate in higher densities close to both natural reefs and manmade structures, such as outfalls (Davis et al. 1982).



Figure 4.11 Photograph of a Tubeworm (*Diopatra* sp.)

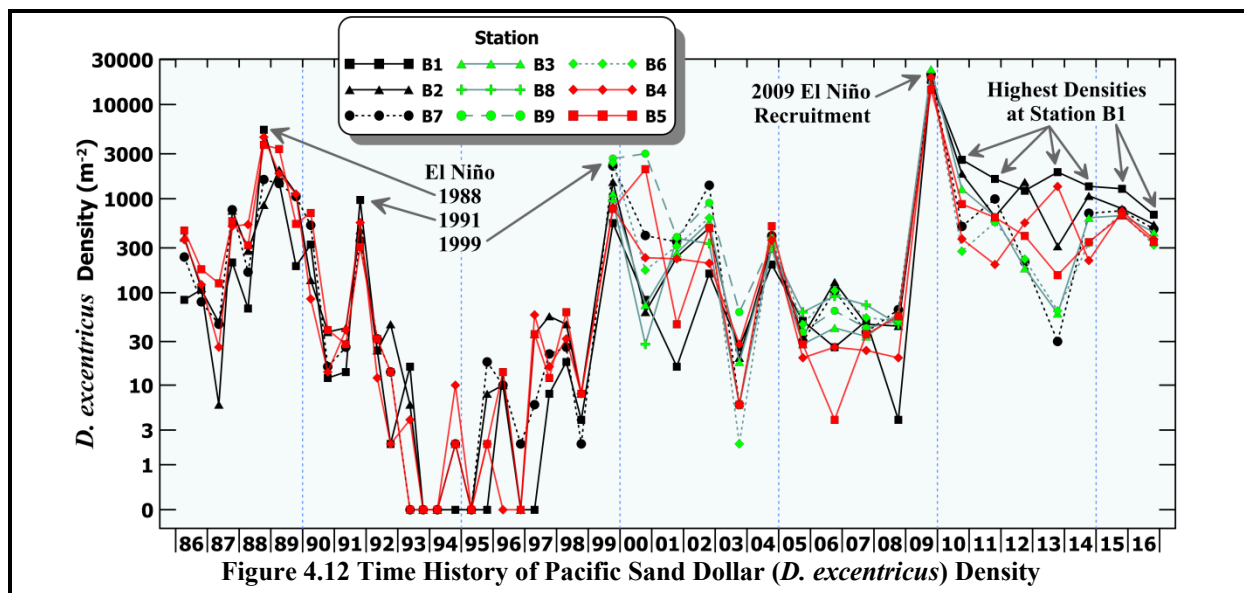
During the October 2016 survey, several small- to medium-sized *D. ornata* tube casings were separated from four of the sediment samples.¹ Some of the casings contained adult worms that were included in the infaunal enumerations while other tubes were empty. The small size and limited occurrence of the tube casings found during the October 2016 survey departed from the more numerous larger specimens found in recent surveys prior to 2015. During 2015, however, no *D. ornata* tube casings were found within the infaunal sediment grab samples, and that general population decline extended into 2016. The decline may be related to the dense bed of mature sand dollars (*D. exentricus*) that disturbed surficial sediment and associated infauna within all the grab samples collected during the 2015 and 2016 surveys (Woodin 1978). In prior surveys, removal of the parchment tubes prior to sieving helped limit the introduction of associated epibionts, but some of these organisms also reside in sediment immediately surrounding the

¹ Refer to the comments in the Sediment-Grab-Sample Collection Record in Table C.1 in Appendix C.

tubes (Wooden 1978), and as a result, their influence was still partially evident in the historical infaunal record. Thus, even though the tubes were removed immediately upon retrieval of the sediment grab samples, those samples were found to contain fauna that were distinct from the communities in all the other replicate samples (MRS 2013–2015).

Sand Dollars

Statistical analyses performed on the 2016 benthic samples also excluded enumerations of Pacific sand dollars (*D. excentricus*). Although sand dollars can be sensitive to pollution, they are excluded from the database prior to statistical analysis because they exhibit extreme population fluctuations from year to year (Figure 4.12), which can potentially mask discharge-related impacts on less-dominant taxa. Additionally, because they tend to reside on the sediment surface, they are not, strictly speaking, infauna, and because their mature size represents a significant fraction of the dimensions of the grab sampler, there is a greater risk of undersampling sand dollar populations as compared to most infauna, which are far smaller. Lastly, because their populations can be extremely patchy, surveys prior to 2016 have found that even closely spaced replicate grab samples contain vastly different abundances, confounding the determination of spatial trends in their distribution across the survey area.



Sand Dollar Biology

Sand dollars form dense beds just seaward of the wave break. At that location, predation is limited because predators, consisting mainly of sea stars, can move over the sand-dollar bed only at significant risk of tumbling into the surf zone with a passing wave (Morin et al. 1985). Consequently, sand-dollar beds migrate shoreward in the summer and seaward in the winter in response to seasonal wave action (Oliver et al. 1980). A winter of strong wave action, frequently associated with El Niño events, may reduce predation, permitting enhanced recruitment of juvenile sand dollars. Specifically, the increased sand-dollar recruitment that occurred in late 1988, 1991, 1999, and 2009 may have been related to El Niño events that occurred around the same times (Figure 4.12).

Sand-dollar beds are often dominated by a single age cohort, indicating that recruitment is episodic and only occasionally successful (Cameron 1980, Oliver et al. 1980, Morin et al. 1985, Cameron and Rumrill 1982). During low-population years, the exclusion of *D. excentricus* specimen counts from the database had little impact on the statistical analyses of benthic populations. This was the case between 1992 and 1998 when average sand dollar densities were less than 50 m⁻². However, because their numbers overwhelmingly dominate the population statistics during and following high-recruitment years, inclusion of sand-dollar enumerations has the potential to mask any influence of the discharge on less-dominant taxa. The populations of taxa that may be sensitive to potential discharge impacts are normally stable over the long-term. As a result, slight changes in their populations arising from localized anthropogenic impacts can be more easily discerned when large and variable sand-dollar counts are excluded from the analyses.

2009 Sand Dollar Recruitment

The extremely successful recruitment event that took place in 2009 resulted in the collection of more than 67,000 individual sand dollars, while only 4,289 other infaunal specimens were collected during that survey (MRS 2010). In the presence of such an extreme sand-dollar population, consistent year-to-year evaluation of infaunal community structure could only be achieved by excluding the large sand dollar population from the diversity indices. Although these juveniles incurred a high mortality rate (>90%) over the following year, the cohort of sand dollars originating from this recruitment event steadily increased in size as they matured (Figure 4.13abcd). Another smaller recruitment event occurred in 2013 and resulted in the presence of two distinct cohorts in that database, one consisting of small, first-year juveniles, and the other consisting of large, mature sand dollars.

However, during 2014 and 2015, the sand dollar population was again comprised solely of adult specimens from the original 2009 cohort. Now, however, their increased physical size was displacing an increasing number of infaunal organisms. The respective number of sand dollars collected during each of those surveys (2,331 and 2,788 individuals) was nearly double the amassed total of all other benthic organisms collected during the 2014 survey (1,334 individuals) and more than triple the number of other infauna collected during the 2015 survey (853 specimens). The sand dollar population declined by 43% between the 2015 and 2016 surveys,¹ and the 1,582 sand dollars collected



Figure 4.13 Comparison of the Sand Dollar cohort in a) 2009, b) 2010, c) 2011, and d) 2012

¹ The most recent sand dollar counts for 2016 exhibited a perceptible decline compared to the two recent prior years along the logarithmic vertical axis in Figure 4.12.

during the 2016 survey coincide with a three-fold increase in the infaunal population to 2,544 specimens.¹ Nevertheless, sand dollars constituted more than one-third of the specimens collected during 2016 and because of their large numbers, sand-dollar counts still altered population statistics, potentially masking important variations in community structure among other infaunal taxa.

In particular, community indices computed with the inclusion of the 2016 *D. excentricus* population (Table B.9 on Page B-12) were noticeably different from indices computed without their inclusion (Table B.10 on the following page). Most indices, such as dominance and diversity, measure the uniformity in abundance among species. When sand dollars are included, the survey-wide diversity indices decrease by 12% and dominance increases by 20%. The infaunal trophic index (*ITI*) is unaffected by the exclusion of sand dollars because they are not members of the feeding guilds quantified by that index. Nevertheless, exclusion of the sand-dollar population enhances the impact evaluation by allowing other less abundant, but potentially more pollution-sensitive taxa to participate in the analyses of community indices other than the *ITI*.

Moreover, the sand-dollar time history does not exhibit a readily apparent relationship to outfall proximity over time. As such, compliance assessments that exclude sand-dollar counts are not hampered by their absence. As with the chemical constituent time histories, any trend in the spatial distribution of sand-dollar populations that might be related to outfall proximity would be evident in the sand-dollar population history (Figure 4.12) as a gradual separation between the time histories for the ZID stations (red), for midfield stations (green), and for stations well away from the diffuser structure (black). Instead, large fluctuations in the overall record of sand-dollar abundance tend to occur equally at all stations. However, there is one notable exception; the spatial distribution of 2009 cohort's abundance was consistently higher at Upcoast Reference Station B1 in all six years following the recruitment event, except in 2012 when it was a close second (black squares on the right in Figure 4.12). For the reasons presented elsewhere in this report, that station is excluded from the compliance analyses.

Other Sand Dollar Influences

As described above, the overwhelming influence of sand dollar variability on infaunal population statistics is easily eliminated by simply excluding their counts prior to the analyses. However, the presence of sand dollars can also strikingly affect the benthic infaunal analysis in two other ways. In contrast to the influence of high numbers on population statistics, these two other influences cannot be mitigated, and within the 2015 database, the overwhelming physical dominance of the numerous large sand dollars severely compromised the ability compare that year's data with prior non-*Dendraster* infaunal analyses, even after their counts were excluded.

First, the presence of numerous large sand dollars within grab samples markedly reduced the sediment volume remaining within the grab samples. Second, their presence altered the infaunal community by introducing parasitic and opportunistic taxa capable of coexisting with them, and by eliminating taxa that normally reside in a less-disturbed benthic environment. The burrowing activity of sand dollars reworks surficial sediments and eliminates delicate suspension-feeding taxa that are considered sensitive to pollution because their populations decline upon contaminant exposure in undisturbed benthic environments. However, these taxa are also highly sensitive to mechanical disturbance of surficial sediments that is unrelated to contaminant content. Similarly, the overwhelming presence of sand dollars also allows the incursion of a few opportunistic taxa that can coexist with sand dollars, and that under other circumstances, would be considered pollution tolerant (Chia 1969; Rhoads and Young 1971; Woodin 1978; Merrill and Hobson 1970; and Smith 1981).

¹ Refer to the first (*D. excentricus*) row and the two Total rows at the bottom of Table B.1 in Appendix B.

Reduced Sediment Volume during the 2014 and 2015 Surveys

The sand dollar population record shown in Figure 4.12 does not lend any insight into these other two sand-dollar influences that affected infaunal analyses of the 2014 and 2015 survey data. The Figure shows that sand dollar density has been relatively constant for the seven surveys following the initial recruitment of vast numbers tiny specimens during the 2009 El Niño event (Figure 4.13a). While the number of sand dollars collected since that survey has remained relatively constant, their size has increased dramatically (Figure 4.13bcd). The associated increase in sand-dollar volume successively displaced greater amounts of sediment within the grab samples. Thus, the amount of sediment actually processed for infaunal organisms steadily decreased with each successive survey and during 2014 and 2015, the presence of numerous, very large, mature sand dollars left little sediment in the grab for sieving offshore. As a result of the smaller sediment volumes, the numbers of infauna were markedly reduced in those years, which dramatically affected every aspect of infaunal community evaluation, and made comparison with data collected in prior years untenable.

Reduced sediment volumes resulting from large sand dollar volumes only became problematic in 2014 and 2015 when much older and larger individual sand dollars were prevalent. Population increases associated with past sand dollar recruitment events (in 1988, 1991, and 1999 in Figure 4.12) generally dissipated within a year or two of the initial recruitment, and before sand dollars fully matured and grew much larger. In addition to predation¹ and other factors, observed declines in sand dollar populations after previous recruitment events may have resulted from the cross-shore movement of the sand dollar bed out of the survey area in response to changes in wave climatology from year to year (Merrill and Hobson 1970).

This was not the case for the 2009 recruitment. The initial population explosion was much larger than the increases seen in prior events, and it produced a populous cohort of surviving individuals that remained in the survey area for the following seven years. Not only was their abundance relatively constant during this seven-year post-recruitment period, but the population was much denser than most prior recruitments. As the large number of maturing sand dollars grew in size, they began to occupy increasingly larger portions of the grab samples collected in each successive survey. In the years immediately following the 2009 recruitment, adequate sediment volumes were collected despite the presence of numerous smaller sand dollars. However, beginning with the 2014 survey, strikingly lower sediment volumes were recovered from the grabs; and during 2015, seafloor sediment occupied only a small fraction of the grab sample. As a result, the infaunal population was severely undersampled during the 2015 survey, which compromised accurate characterization of the community and precluded reliable comparison with the historical database.

Although sand dollars collected during the 2016 survey were still uniformly large, their numbers declined enough to allow more-representative sediment sample-volumes to be collected within all the grabs. Thus, the sediment sample-volume artifacts seen in the prior two surveys were not as apparent in the population statistics described in this report.

Taxonomic Influence

While the 2016 decline in sand-dollar counts ameliorated the some of the issues associated with insufficient sediment-sample volume, the presence of numerous large sand dollars still exerted a profound

¹ Surveys conducted in the year immediately following prior recruitment events encountered increased numbers of sea stars, usually ochre stars (*Pisaster ochraceus*), which are a primary sand-dollar predator. In contrast, few sea stars were encountered in the benthic surveys conducted after the 2009 recruitment, and this lack of predation may have contributed to the observed longevity of the sand dollar population. The recent absence of sea stars in the survey area may be related to the outbreak of sea star wasting disease, which particularly impacts ochre stars.

taxonomic influence on the resident infaunal community. In addition to their impact on sample volume, the presence of a large swath of mature sand dollars within the survey area directly affects infaunal community composition by introducing opportunistic taxa that are capable of coexisting with them, and by eliminating suspension feeders that normally reside on the surface of an undisturbed benthos. Specifically, sand dollar predation on small crustaceans, such as amphipods and isopods, virtually eliminated some of these normally prevalent taxa from the 2015 and 2016 database. Other impacts on the infaunal community arose indirectly through enhanced competition for food and space. Sand dollars can be both deposit and suspension feeders, and when little sediment is available, or has low organic content, they survive by suspension feeding (Timko 1976). Most other infauna do not have the same trophic flexibility and losses due to competition with the fully mature cohort of sand dollars, undoubtedly contributed to the observed declines in species richness and diversity within the remaining infaunal community, as well as anomalies in other community properties described below.

4.3.2 Indigenous Infaunal Community

The type and abundance of benthic organisms can directly reflect the quality of the marine sediments where they live. Infaunal communities residing in degraded sediments tend to exhibit low species richness and diversity because they are populated by only a few opportunistic taxa that feed on detritus and thrive in sediments that are high in subsurface organics. In contrast, clean sediments are populated by diverse assemblages of organisms that include filter feeders, which extract nutrients from suspended particulates. These suspension feeders are sensitive to excessive organic-particulate loads such as those that can be discharged from low-performance treatment works. Historically, the widespread presence of suspension feeders around the MBCSD outfall indicates that the discharge was not adversely affecting them. One such suspension feeder is the club-tipped anemone (*C. californica*) that carpets the diffuser ports (see the photograph in Figure 2.2 on Page 2-4). Their presence is indicative of the benign nature of the suspended sediments discharged by the treatment plant.

In this and past reports, the overall health of the benthic community within Estero Bay is characterized by the principal population parameters computed from samples collected at each station over the duration of the monitoring program. The community parameters provide a succinct description of the abundance, diversity, richness, and trophic (feeding) structure of benthic biota. Figure 4.14 on Page 4-38 shows the temporal variability in mean community parameters that were computed for each of the nine historic benthic monitoring stations, seven of which have been monitored continuously, including during 2016. Each major category of community parameter is discussed in the sections that follow.

Abundance

Population density, shown in Figure 4.14a, measures the total abundance, or number of organisms, within a 1-m² area of the seafloor. For the reasons described above, infaunal density and the other infaunal community parameters analyzed here exclude sand-dollar counts. Although population density is the most basic single measure of a community of organisms, it lacks valuable insight as to how the abundance is distributed among the differing taxa. Because of this deficiency, it is not a useful ecological measure by itself. Nonetheless, major declines in organism density can occur after a rapid increase in sediment contamination. This initial decline is often followed by a marked increase in total abundance as high numbers of opportunistic species repopulate the degraded environment.

Throughout the monitoring program history, the number of organisms collected at each station has fluctuated enormously. Although the range in abundance covers an order of magnitude, differences in abundance among individual stations within any given survey were generally smaller. Namely, all the stations tend to track the temporal fluctuations as a group, and spatial differences among stations within a

given survey are generally no larger than the wide temporal fluctuations between surveys. The relatively large temporal fluctuation places perceived spatial differences within a given survey into context; namely, if an ecosystem is able to accommodate seasonal and interannual population fluctuations with amplitudes exceeding spatial gradients, then the spatial differences observed during a given survey are probably not environmentally significant.

In that regard, prior to 1999 when seasonal survey data was available, the largest population changes occurred on a seasonal basis. Post-summer populations (those sampled during the October survey) were generally a factor of three higher than the post-winter populations. The reduced winter population, which was only measured prior to 1999, is a response to harsher oceanographic conditions when storms generate intense wave-induced currents that scour the bottom near the 15-m water depth of the outfall. Post-summer populations reflect recruitment of the more delicate annelid worms that can only survive during quiescent benthic-flow regimes. The absence of winter surveys after 1998 accounts for the uniformly higher densities that are apparent in Figure 4.14a for most of the last half of the monitoring record, at least until after the 2009 sand-dollar recruitment.

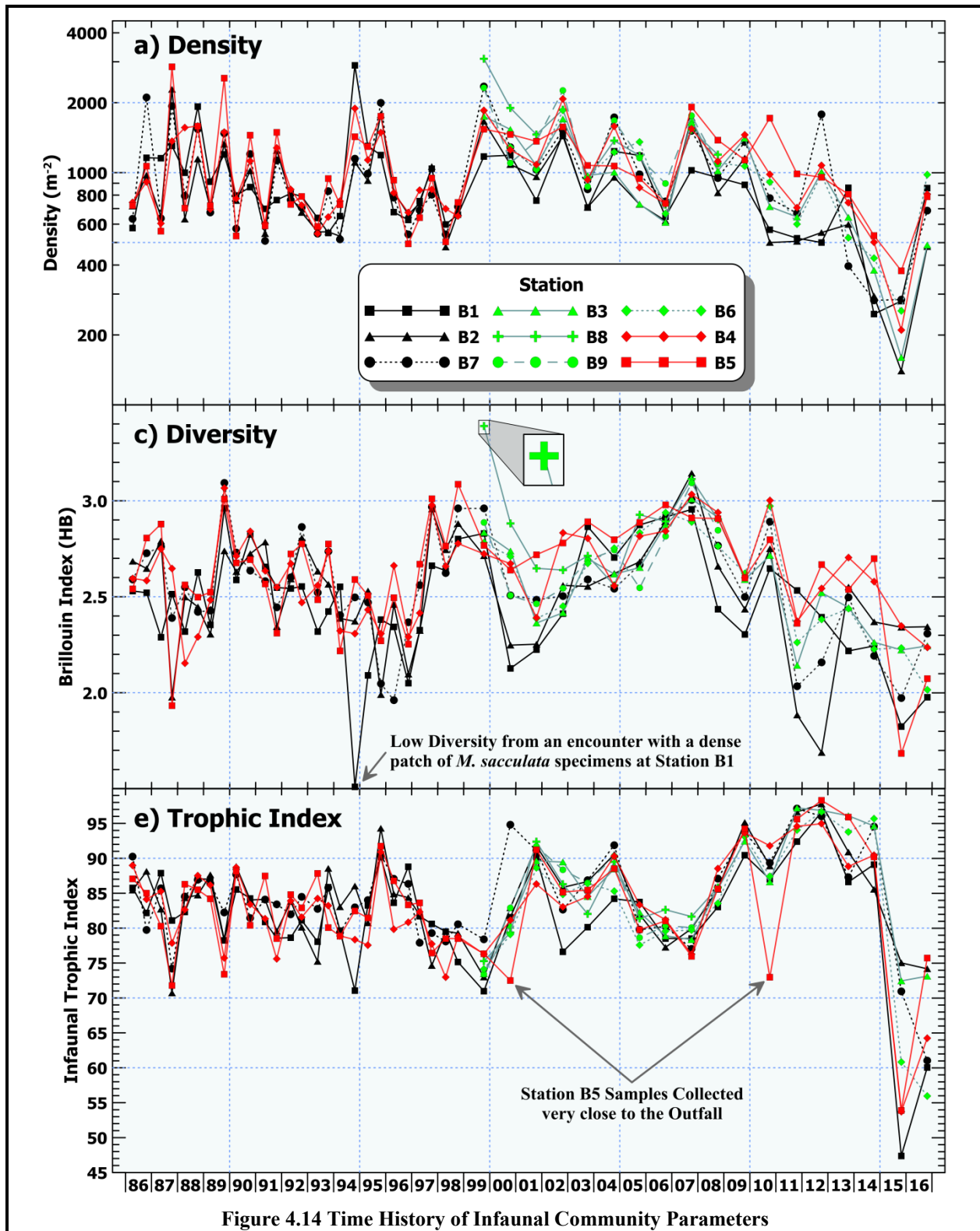
Prior to 2009, small spatial gradients that may be apparent in density within a given survey tend to be obscured by the large temporal fluctuations. In particular, infaunal abundance at stations near the outfall (shown in red) were not consistently lower than the density at distant stations (shown in black), and this lack of spatial distinction has remained constant throughout the monitoring program. It demonstrates that any potential accumulation of sediment chemicals from the effluent discharge over the last three decades has had no discernible impact on infaunal abundance, particularly when compared to large seasonal and interannual population fluctuations.

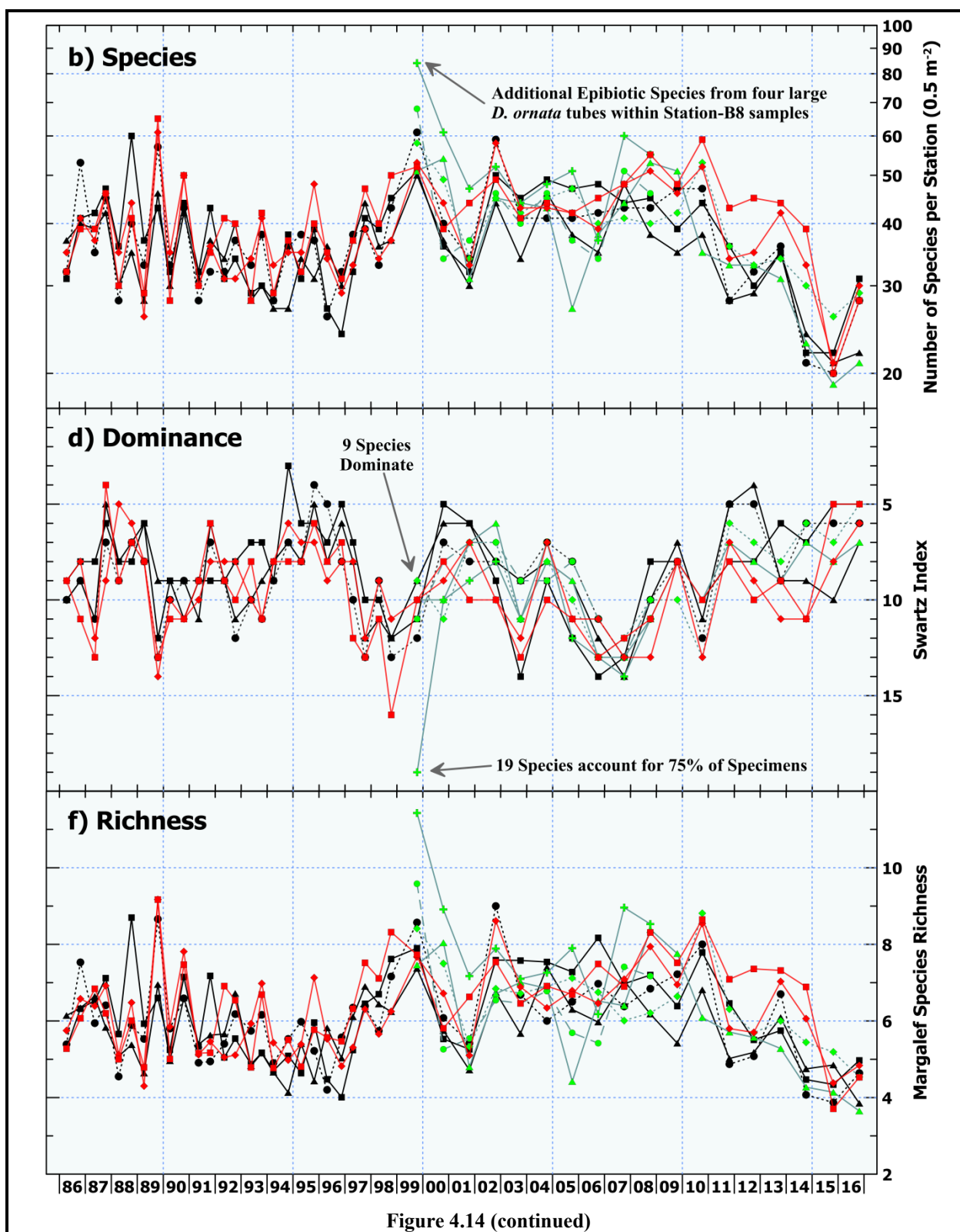
2009 Sand Dollar Recruitment

Following the major sand-dollar recruitment in 2009, overall infaunal densities (excluding sand dollars) began to decline. In 2010, infaunal density dropped to levels comparable to those of the post-winter surveys conducted prior to 1999. Infaunal densities remained at about these levels through 2013, probably in response to competition for food and space, and to predation by the large population of sand dollars. Additionally, during the seven surveys conducted after the 2009 recruitment, sand-dollar abundance at Upcoast Reference Station B1 was almost always the highest (black squares in Figure 4.12) and accordingly, the density of other infaunal organisms at that station were often lower than at other stations (black squares in Figure 4.14a).

During 2014 and 2015, infaunal densities precipitously declined and reached the lowest densities recorded in the 31-year database. This dramatic recent decline in the abundance of infauna cannot be easily ascribed to predation or competition for resources, particularly considering that the numbers of sand dollars in 2014 and 2015 were comparable to prior post-recruitment years (Figure 4.12). Instead, as previously discussed, the sharp decline in infaunal populations was partially an artifact of undersampling caused by the physical presence of the numerous mature sand dollars within the grab samples. The sheer volume of sand dollars displaced the sediment normally collected with grab samples. Consistent sediment-sample volumes are necessary for accurate intercomparison of infaunal abundances determined from the enumeration of taxa within individual grab samples. For that reason, infaunal sampling protocols require a grab penetration depth¹ of at least 7 cm to ensure that a sufficient volume of sediment is collected to assess the indigenous community adequately and consistently. Nevertheless, when large sand dollars occupy most of the grab volume, infaunal populations within the remaining sediment are undersampled, leading to an under-reported density.

¹ Refer to the Penetration Depth measurements for the 2016 samples in Table C.1 in Appendix C.





The decline in sand-dollar abundance during the 2016 survey appears to have largely ameliorated the sample-size problems experienced during the 2014 and 2015 surveys. Infaunal populations rebounded sharply at all stations and reached levels of 500-to-1000 m⁻², which were comparable to those of the 2010-to-2013 post-recruitment period. However, they were still below those of the pre-recruitment decade of post-summer surveys, suggesting that the sand-dollar taxonomic influence was still present during the 2016 survey.

Number of Species

The total number of species is another measure of the overall health of benthic communities (Figure 4.14b). Everything else being equal, infaunal communities respond to environmental stresses by becoming less diverse, as reflected by a substantial reduction in the number of species. However, the number of species encountered in a sample is directly related to the number of organisms in the sample, i.e., it is inherently related to infaunal abundance. Because rare species are generally undersampled, the number of species automatically increases in samples that contain more organisms. Thus, it is difficult to tell if a decreased number of species is due to an actual decline in biodiversity, or simply due to decreased sample sizes (abundance). Irrespective of the influence of sample size, however, reductions in species counts could suggest a decline in diversity and degradation in the marine environment.

Because of its interrelationship with population density, temporal fluctuations in species counts (Figure 4.14b) tend to track the fluctuations in abundance (Figure 4.14a). Accordingly, at least some of the variability in the numbers of species at individual stations within the MBCSD database is an artifact of changes in the sample size (abundance) rather than purely an increase in species richness or diversity. As with the density time histories, species counts in post-winter surveys, which were conducted prior to 1999, were generally lower than post-summer counts. Additionally, in conjunction with density, the numbers of species encountered in samples collected in the last seven years were perceptibly lower than those of the prior decade, and were markedly lower in 2014 and 2015, particularly at the stations containing numerous large sand dollars.

In general, however, there is no obvious long-term spatial gradient in numbers of species suggesting that sedimentary conditions near the outfall have been preferentially degraded. Specifically, there is no obvious divergence in the colors over time, where species counts at stations near the outfall, shown in red, steadily decline relative to stations far from the outfall (black). As with abundance, spatial differences in within the 2016 species dataset itself were limited, and comparable to seasonal and interannual fluctuations of past surveys, even though the overall number of species had declined to the lowest levels on record during the two prior years (2014 and 2015).

Beginning in 1996, there was a brief three-year increase in the overall number of species encountered in the MBCSD samples. This trend culminated in 1999, when 84 species were collected at Station B8 alone (green line with crosses). However, this increase was largely due to the increased prevalence of ornate tubeworms (*D. ornata*) within some of the samples. For example, between 1990 and 1996, only one, and usually no *D. ornata* specimens were collected during any given survey. However, in the post-winter surveys of both 1997 and 1998, two ornate tubeworms were encountered. In 1999, four of these tubeworms were found at Station B8 alone. These tubes were not removed prior to sieving as they are now, and as discussed previously, the inclusion of numerous epibionts associated with their tube casings contributed to the steady increase in species counts, culminating in the unusually large number of species recorded in 1999.

Reduced Sediment Volume in 2014 and 2015

As with density, the historically low numbers of species reported in 2014 and 2015 could easily be explained by the reduced sediment volume caused by the presence of large sand dollars. Based on rarefaction analysis,¹ the observed decrease from 40 to 20 species would have resulted from a five-fold reduction in sediment volume. Based on anecdotal visual observations of the grab samples collected during the 2015 survey, sand dollars typically filled about 80% of the grab volume prior to sieving. Thus, their displacement of benthic sediments could easily explain the observed decrease in both infaunal density and average species. Other field studies that quantitatively accounted for differences in sediment volume within sand dollar beds also found reduced diversity and numbers of infaunal species that were related to the limited sample sizes within the beds (Smith 1981).

Diversity

A large number of interrelated diversity indices have been developed in an attempt to account for the inherent relationship between the number of species and sample size or abundance. Two common indices are the Shannon Diversity (H') and the Brillouin index (HB) (Appendix B.1.1). Both exhibit nearly identical time histories in the MBCSD database, so only the Brillouin index is shown in Figure 4.14c.

Ostensibly, diversity should decrease in a degraded benthos, although significant reductions can also occur in the absence of anthropogenic stresses. Nevertheless, within the MBCSD database, temporal trends in diversity do not exhibit any evidence of a deteriorating benthic environment near the outfall; namely, a steadily decreasing diversity at ZID stations relative to distant stations. On the contrary, the highest diversity has often been observed at one of the two stations located closest to the outfall (shown in red).

Nevertheless, because of the aforementioned taxonomic influence from sand dollars after 2010, diversity measures have been of questionable value for assessing potential impacts from the discharge over the following six years. During that time, diversity dropped below 2.5 at most stations and, with a within-survey range near 0.5, the diversity index exhibited much higher variability among stations than seen in prior years. Furthermore, the wide excursions seen at individual stations during recent years could not be readily ascribed to other aspects of the samples collected those stations, such as numbers of sand dollars, grab penetration depth, or presence of ornate tubeworms. Instead, the large observed diversity fluctuations were probably in response to varying levels of competition with sand dollars for space and food.

During and prior to the 2009 sand-dollar recruitment, and in the year immediately afterward, diversity measured at individual stations within a given survey were fairly similar, and when isolated large excursions were observed, their cause could be easily identified. For example, the two most prominent outliers occurred during the 1994 and 1999 surveys. The markedly high diversity (3.39) that was measured at nearfield Station B8 in 1999 was an artifact of the enumeration of additional epibionts residing on the casings of ornate tubeworms. As discussed previously, the presence of large numbers of epibiont species substantially increased both the overall abundance, and number of species counted at this station during the 1999 survey (Figure 4.14ab). Not surprisingly, the diversity reported at that station was also markedly elevated (green cross in the inset of Figure 4.14c)

The second unusual outlier is the very low diversity (1.51) that was found at Reference Station B1 during the October 1994 survey (black square ■ in Figure 4.14c). As with outlier described above, there were an unusually high number of organisms enumerated in the samples collected at this station in 1994 (Figure 4.14a). However, in this case, the increased abundance was associated with a single species, the

¹ Rarefaction (Hurlbert 1971) provides an estimate of species counts as a function of sample size.

spioniform polychaete worm *Magelona sacculata*. Nearly 1,000 specimens were collected at Station B1 during the October 1994 survey. Because of the numeric dominance of this one taxon, the computed diversity at this station was the lowest measured in the 31-year database.

Dominance

The Swartz species dominance index differs from the community indices discussed previously because it does not make an *a priori* assumption concerning how individuals are distributed among species. Because it is non-parametric, it is less sensitive to the limitations that plague other community indices. The index is defined as the total number of species accounting for 75% of the individual organisms collected. As such, it represents an inverted measure, in which increased dominance is associated with lower values of the Swartz index.¹ Increased dominance is expected in degraded benthic communities where a few opportunistic, pollution-tolerant taxa abound in the absence of a wider array of pollution-sensitive organisms. As with the other community parameters, Figure 4.14d shows that differences in dominance among stations from a given survey were generally comparable to, or smaller than temporal fluctuations, at least prior to 2011.

After that, however, the dominance among individual stations from individual surveys ranged widely. Again, this reflects the varying taxonomic influence of the large sand dollar populations on the infaunal populations within individual replicate grab samples. At the stations with the highest dominance, only four-to-six species shared numerical superiority, which is among the highest measured in the 31-year database. However, the wide excursions in dominance that occurred after 2010 are unrelated to the environmental quality within individual benthic samples. A potentially degraded environment at a particular location would be reflected in a time history where that one station's dominance is consistently higher than that of the other stations. Instead, the station with the highest dominance varied randomly from year-to-year.

This was not the case prior to 2010, when outliers were easily ascribed to conditions within individual samples; in particular, the Station-B8 samples collected during 1999 that contained epibionts from multiple ornate tubeworms. The presence of the tubeworms resulted in the lowest historical dominance by far, and allowed quantification of the actual number species that the tubeworm casings added to the samples. Within the Station-B8 samples, nineteen species accounted for 75% of the specimens collected, while other stations sampled during the 1999 survey had as few as nine species. Thus, up to 10 of the 19 dominant species that were found at Station B8 during 1999 were epibionts associated with the presence of ornate tubeworms. The exclusion of these tubeworms from subsequent surveys resulted in more stable and representative values of the Swartz index, at least until 2011.

Clean-Sediment Feeding Guilds

The Infaunal Trophic Index (*ITI*) (Word 1978, 1980, 1990) compares the abundance of four soft-bottom assemblages that are distinguished by their feeding behavior. Because sensitivity or tolerance to organic enrichment differs among the four groups, shifts in the dominance of individual feeding guilds, as reflected by a change in the Trophic Index, can be indicative of a benthic environment degraded by excess organic material. The *ITI* ranges between 0 and 100. When species in Group I (suspension feeders) and Group II (surface-detritus feeders) dominate, index values are above 58, indicating that sediments are relatively clean, or at least devoid of organic material. Lower infaunal indices occur when populations of species in Group III (surface deposit feeders) and Group IV (subsurface detritus feeders) are prevalent.

¹ Note that the vertical axis of the Swartz index plot in Figure 4.14d is inverted so that values that are located higher in the plot represent increased dominance.

Their increased presence is assumed to result when sediments are rich in the organic material that acts as a food source for these types of infaunal organisms.

As with the Swartz dominance index described above, the *ITI* differs fundamentally from the other interrelated community indices and as a result, its time series (Figure 4.14e) does not closely track the excursions in other indices (Figure 4.14abcf). Its emphasis on feeding behavior provides a somewhat independent assessment of the health of the infaunal community. With the exception of 2015 and 2016, the *ITI* has been exceptionally high throughout the MBCSD monitoring program; often exceeding 80, and after 2008, exceeding 90 (Figure 4.14e). These very high index values reflect a community almost entirely dominated by suspension-feeding organisms, with few organisms that depend on organic material within the substrate to survive. Prior to 2015, the exceptionally clean sediments of Estero Bay have always been documented with a trophic index well above the critical *ITI* of 58, below which organically enriched benthic conditions are potentially responsible for the composition of the infaunal community. In fact, trophic indices never dropped below 70 at any of the MBCSD benthic stations sampled during the 29 years prior to 2015, confirming the consistent dominance of suspension and surface-detritus feeders within the exceptionally pristine sedimentary environment offshore Morro Bay and Cayucos.

Moreover, there is no consistent pattern of organic degradation close to the outfall evident in the *ITI* time series. Increasing effluent-related degradation due to the deposition of organic particulates around the outfall would appear as a gradual separation in colors over time, with stations closest to the diffuser structure (shown in red) eventually having the lowest *ITI*. This would be followed by an increasingly higher *ITI* at nearfield stations, shown in green, with distant stations, shown in black, eventually having the highest *ITIs*. Instead, as with the other community indices, and with the chemical concentrations measured within benthic sediments, there is no consistent pattern of color separation over time. Even during 2015 and 2016, after the *ITI* declined precipitously, no spatially consistent separation in color is apparent that would indicate a decline that was preferentially greater at the ZID stations.

Pre-2015 Outliers

Before addressing the striking *ITI* decline in 2015 and 2016, two other previous outliers warrant discussion because they reflected a reduced *ITI* at ZID Station B5 that stood-out from the other *ITIs* reported in the same year at other stations (red squares in 2000 and 2010 in Figure 4.14e). Although these two *ITIs* were not particularly low compared to *ITIs* reported in some other years, under certain circumstances, they could indicate short-term habitat degradation from the discharge of organic particulates within wastewater. Detailed analysis in Section 4.4 of MRS (2011) did, in fact, identify the outfall as the source of organic material that caused a marked increase in subsurface detritus feeders at that particular sampling station. However, instead of wastewater discharge, the source of the organic loading was the debris field that exists immediately adjacent to the outfall structure. These two instances of anomalously low *ITIs* arose because Station-B5 grab samples were inadvertently collected extremely close to the diffuser structure during both the 2000 and 2010 surveys. In addition to their low *ITIs*, the sediment samples contained significant amounts of shell hash and surficial debris that made them visually and volumetrically distinguishable from other samples collected slightly farther from the diffuser at other stations during those surveys.

Significant amounts of organic debris accumulate in the sediments next to seafloor structures like the outfall pipe, as epifauna die and slough off the structure. In the case of the outfall, debris is also purposely dislodged by divers cleaning the diffusers, or accidentally dislodged when the grab unintentionally collides with the structure. Like the shell mounds below offshore platforms, the accumulation of organic debris can be quite large but is restricted to the seafloor immediately adjacent to the structure. Consequently, the outfall structure's debris field is rarely encountered except when grab samples are

collected within a few meters of the outfall. In addition to the organic enrichment, these structural debris fields contain highly disturbed sediments that support a unique community of benthic organisms that also happen to have an affinity for high concentrations of organic material, which weighs heavily in the reduction of the *ITI*.

ITI Decline in 2015 and 2016

As with the two anomalously low *ITIs* described above, the abrupt *ITI* decrease observed at all stations in 2015 and 2016 cannot be ascribed to the discharge of organic particulates within wastewater. As with the other community indices, which are somewhat independent of the *ITI*, the *ITIs* in both years were highly variable among replicate samples collected at each individual station. The resulting station *ITIs* spanned a remarkably wide range; as much as 28 compared to prior years, which were typically 10 (Figure 4.14e). Within the highly variable *ITI* results from 2015 and 2016, there was no evidence a consistent gradient of decreasing *ITI* with increasing outfall proximity; namely, lower *ITIs* shown in red versus those shown in black.

More importantly, the unmistakable overall *ITI* decrease in 2015 and 2016 was not solely due to a marked increase in the populations of subsurface detritus feeders (Group IV), which would suggest an influx of opportunistic organisms. Instead, the decline also resulted from a large reduction in the population of suspension-feeding organisms. The lowest *ITIs* measured during 2015 and 2016 contrasted sharply with the *ITIs* measured in surveys conducted just a few years before in 2011 and 2012, when the *ITIs* were the highest ever observed in the entire 31-year benthic monitoring program (Figure 4.14e). With *ITIs* as high as 98, the infaunal community at that time consisted almost entirely of suspension feeders (Group I) residing in sediments largely devoid of any organic content whatsoever.

If the MBCSD effluent discharge had caused an accelerated accumulation of organic particulates around the outfall between 2012 and 2015, the *ITI* reduction would have definitely been more apparent at the ZID stations during at least some of the intervening years. Considering the large magnitude of the overall *ITI* decline, the profound transition from an infaunal community of effectively all suspension feeders to exclusively deposit and detritus feeders would have been plainly evident at the ZID stations. On the contrary, the lowest the *ITIs* during those years were regularly measured at the distant reference Station B1, indicating that organic enrichment was occurring regionally, and was not due to an inhospitable benthic environment created by a sudden increase in sediment organic content near the outfall. Such a localized buildup of organic materials would have been evident in the time series of all three of the organic loading constituents (Figure 4.9c). The localized buildup would have been apparent as a statistically significant increase at the ZID stations relative to prior measurements and measurements at other stations during 2015 and 2016.

As previously discussed, the increased sand-dollar size was at least partially responsible for the perceived taxonomic changes within the infaunal communities during recent years. Their growth increase resulted in an attendant reduction in the volume of surficial sediment collected in the grab samples in 2015, and to a lesser extent in 2016, compared to all prior years. Undersampling causes a disproportionate reduction in the number of organisms collected from the most populous taxonomic group, which for past MBCSD benthic surveys has been the assemblage of active suspension-feeding Gammarid amphipods. These organisms are small shrimp-like crustaceans (Figure 4.15). Specifically, *Foxiphalus xiximeus*, *Majoxiphalus major*, *Rhepoxynius abronius*, and *Rhepoxynius menziesi* have dominated the infauna collected in the MBCSD monitoring program as a whole, and because they are all members of Trophic Group I, their near elimination in 2015 and 2016 weighed heavily on the observed decline in the *ITI*. For example, in 2012, 156 *F. xiximeus* and *M. major* specimens were collected, while in 2015, only eight of

these organisms were found. Similarly, the number of *R. menziesi* declined by an order of magnitude in 2015, and *R. abronius* disappeared entirely.

However, undersampling alone, cannot account for the sand-dollar-induced decline in these gammarid populations. These amphipods are known to be extremely sensitive to pollution (Pearson and Rosenberg, 1978) and because of this, are routinely used in bioassay testing of marine sediments (ASTM, 1991; USEPA-USACOE, 1991, 1993). However, as described in Section 4.2, there was no material change in sediment quality during 2015 and 2016, and no new known source of pollution within Estero Bay. Instead, the extraordinary decline in the amphipod population was probably caused by habitat disturbance, competition for space and food, and predation by the numerous large sand dollars that were present in 2015 and 2016.

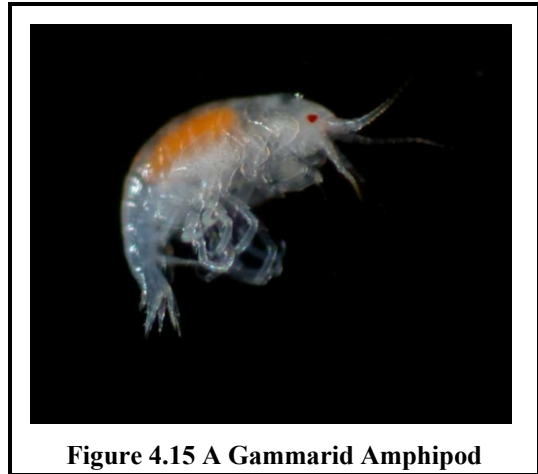


Figure 4.15 A Gammarid Amphipod

While sensitive to sediment pollution, gammarid amphipods are also sensitive to bioturbation caused by burrowing sand dollars, which can result in changes in surficial grain size and moisture content that cause amphipod mortality (DeWitt et al. 1988). In contrast to prior years, when, except for a few sand dollars, a generally smooth sediment surface was observed in grab samples, the surface of grab samples collected in 2015 and 2016 consisted almost entirely of large sand dollars with little or no surficial sediment visible. The high rugosity of the seafloor during those years mechanistically limits the mobility of organisms that normally forage for particulate material on or near the sediment surface.

In addition to direct competition for space on the sediment surface, the sand dollars increasingly compete for food with suspension feeders as their density increases (Fodrie et al. 2007). At low density, sand dollars are predominately deposit-feeders. Thus, in prior years, the physically smaller sand dollars produced lower density beds, and in their deposit-feeding mode, preferentially competed with Group II and IV detritus-feeding organisms rather than suspension feeders. At least in part, this competition was responsible for the lower population of detritus feeders that were observed during, and in the five years following the 2009 sand-dollar recruitment event. The low numbers of detritus feeders resulted in the exceptionally high *ITI* observed during that period.

However, with the marked increase in the density of sand dollars in 2015 and 2016, suspension feeding became the sand dollar's preferred feeding mode. Reduced competition with detritus feeders resulted in the small observed increase in numbers that brought Group II and IV populations closer to historical levels, when sand dollars were less prevalent. In contrast, increased competition with Group I suspension feeders contributed to their near elimination in the 2015 and 2016 sediment samples. The combined effect of a near-normal deposit feeding population and a severe reduction in suspension feeders caused the *ITIs* to drop to a historic low during 2015 and 2016.

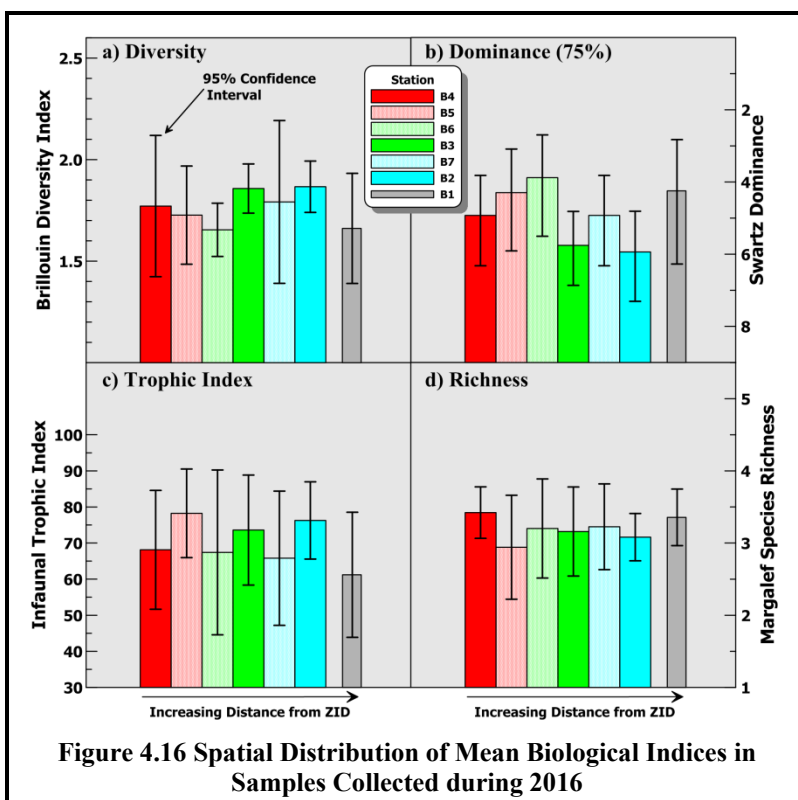
In addition to competition for food, small crustaceans, such as gammarid amphipods, can comprise the majority of sand dollar prey when the sand dollars are in their deposit-feeding mode (Timko 1976). Thus, the population of gammarid amphipods, which are highly influential in the determination of the *ITI*, can be further reduced because of predation by adult sand dollars. While it is unclear what combination of these potential interactions between infauna and the sand dollar bed was responsible for the observed *ITI* decline in 2015 and 2016, it is clear that the presence of numerous large sand dollars was directly

responsible for the low *ITI*, rather than some new pollutant source within Estero Bay sediments, particularly one related to the MBCSD effluent discharge.

4.3.3 Infaunal Diversity Unrelated to Outfall Proximity

The previous section documented the overall health of the infaunal community surrounding the outfall using a 31-year record of infaunal community indices. This section investigates whether spatial gradients related to outfall proximity exist in the infaunal data collected during 2016 alone. As described in Appendix B.1, four of the biological indices are more independent and informative than the other indices. They include the Brillouin Diversity Index, the Swartz Dominance Index, the Margalef Richness Index, and the *ITI*. The 2016 spatial distribution of average values for these four indices is shown in Figure 4.16.¹

The bars shown in Figure 4.16 are arranged in order of increasing distance from the diffuser structure. The red-colored bars on the far left are associated with the ZID stations located closest to the diffuser and within 24 m of a diffuser port. The two green bars in the middle represent the nearfield stations that lie between 36 m and 65 m from the closest portion of the diffuser structure. The two blue bars represent the two mid-field stations that are between 118 m and 124 m from the diffuser. The thinner and separate grey bar on the far right represents the distant reference Station B1 that lies more than 1 km from the discharge. As with the benthic chemistry samples collected at this distant station, its benthic community is unrepresentative of the community found within the survey area surrounding the outfall, and therefore is excluded from compliance analyses.



Impacts to infaunal communities from the discharge would be reflected in the bar chart as a spatial gradient of steadily increasing diversity, *ITI*, and richness from left to right (Figure 4.16acd). In contrast, dominance would be expected to decrease with increasing distance from the outfall (Figure 4.16b). However, none of these spatial gradients is unequivocally apparent in the Figure. Even if slight trends were apparent, they would not be statistically meaningful considering the wide-ranging scatter among individual replicates collected at each individual station, which are reflected by the confidence intervals shown for each bar.

¹ Note that the values plotted in the time histories of Figure 4.14 are station totals, and differ from the averages among replicate samples plotted in Figure 4.16 for 2016.

The confidence intervals themselves provide a quantitative evaluation of the significance of spatial differences among the station averages of infaunal indices. Specifically, they provide a statistical test to determine whether the average value at a given station is significantly different from that of another station, irrespective of their distance from the outfall.¹ The height of the bars in Figure 4.16 reflects the average index computed from the five replicate samples collected at each station. The 95% confidence intervals surrounding those averages quantify the variability among replicate samples collected at each individual station. As such, the confidence intervals reflect the inherent uncertainty in determining the average station values. Thus, if the average at one station is encompassed within the confidence interval at another station (or vice versa), then no statistical significance can be ascribed to any perceived difference between the averages at those two stations, at least with any degree of confidence (viz., a 95% confidence level, $p \leq 0.05$).

Visual inspection of Figure 4.16 indicates that the differences in average indices between most pairs of stations were encompassed by their associated confidence intervals. Of the 60 pairwise comparisons, only four pairs of average station indices were found to be significantly different at the 95% confidence level.² None of the station pairs of average *ITI* and richness was significantly different from one another (Figure 4.16cd). However, the average diversity appears to be significantly lower at nearfield Station B6 (light green bar in Figure 4.16a) than at near- and mid-field Stations B2 and B3 (solid blue and green bars).³ Similarly, the average dominance appears to be higher at Station B6 compared to the same two northern stations (Figure 4.16b). The multivariate analyses described in Section 4.3.5 reveals that Replicate Sample B6R1 contained a highly anomalous infaunal assemblage with reduced diversity and increased dominance that undoubtedly influenced the computation of average community indices at that station and led to the perceived difference from indices at a few other stations.

Regardless, the potentially significant differences found in comparisons between the mean indices at a few pairs of stations do not necessarily indicate the presence of a gradient related to outfall proximity. In fact, the mean diversity does not appear to steadily increase, and mean dominance to decrease with increasing distance from the outfall. Consequently, if weak spatial differences do exist, they may be unrelated to outfall proximity. In particular, the far more robust multivariate analyses presented in Section 4.3.5 indicates that very subtle spatial differences in the infaunal community occur along a north-south gradient within the survey area. Even in those analyses, however, the spatial differences are difficult to discern because any differences in the infauna between stations is largely overwhelmed by the wide variability among replicate samples collected at any given station.

In that regard, the hypothesis tests on infaunal indices that were described in this section found significant differences in only four (7%) of the 60 pairwise tests. However, by the definition of a 95% confidence interval, one-in-twenty tests (5%) are likely to result in a finding of significance by chance alone. Specifically, the test statistic was not adjusted for multiple comparisons. For example, a Bonferroni adjustment for five comparisons⁴ increases the likelihood of incorrectly finding a significant difference from one-in-twenty to one-in-five. After applying a Bonferroni correction for multiple comparisons, none

¹ This is equivalent to an unpaired two-sample Student *t*-test of the null hypothesis that there is no difference in mean values between stations at the 95% confidence level assuming unequal variances and a two-tailed distribution.

² Station B1 was excluded from the pairwise comparisons

³ The confidence limits surrounding the average for the third (light green) bar do not encompass with averages for the fourth and sixth bars, and vice-versa in Figure 4.16a.

⁴ A repeated comparison of the average of a given index at one station against the averages of that index at the five remaining stations within the survey area (excluding Station B1)

of the perceived differences in station averages was found to be statistically significant at the multi-comparison equivalent of a 95% confidence interval.

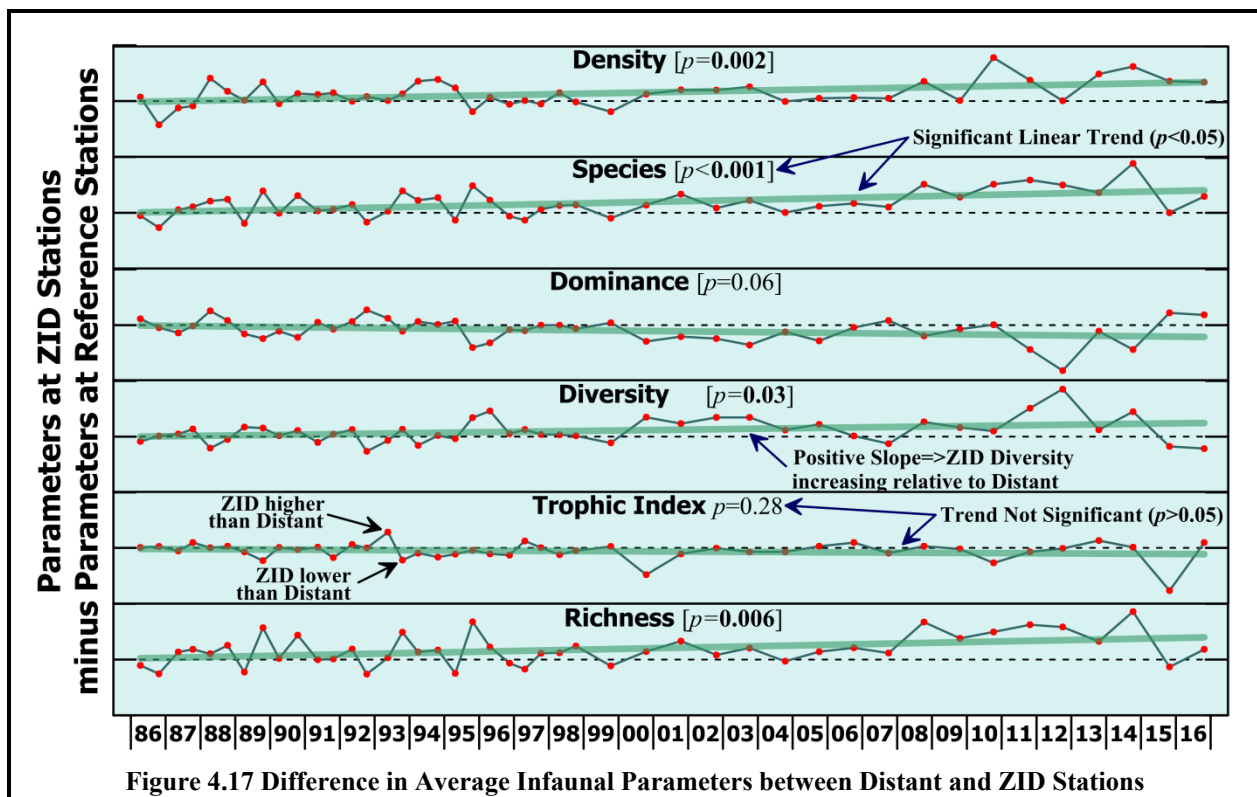
4.3.4 Spatiotemporal Infaunal Trends Unrelated to Outfall Proximity

As with the chemical constituents within northern Estero Bay sediments, subtle spatiotemporal trends in the infaunal community can be difficult to discern by visual inspection of the time series shown in Figure 4.14. The parameters exhibit wide fluctuations between seasons and years. Compared to these large temporal oscillations, the spatial differences among samples collected at a given time (i.e., within a given survey) were comparatively small. In other words, the stations generally tracked one another through time. This indicates that changes in infaunal communities that arise from regional influences throughout Estero Bay are far larger than those induced by any localized effects. Therefore, within a given survey, spatial differences among infaunal parameters can be difficult to differentiate, and are likely to be confounded by larger natural differences in the sedimentary environment, as well as other zoogeographic influences.

The limitations associated with purely spatial or purely temporal approaches to impact evaluation are largely eliminated through application of the same statistical analysis technique used in Section 4.2.3. It is based on tests for nonparallel trends at impact and reference sites (Coats et al. 1999, Skalski et al. 2001). In this application, these parallelism statistical tests are capable of discerning subtle temporal trends in the difference between average indices at stations located near the outfall, and those that are distant from the outfall, even though the indices as a whole fluctuate widely from survey to survey. As described previously, discharge-related spatiotemporal changes would appear in Figure 4.14 as a gradual separation of the colors as contaminants accumulate near the outfall and affect the resident organisms. However, because of the large coherent fluctuations that occur over time, subtle spatiotemporal changes in community indices may be present but not visually apparent in that kind of presentation. Taking the difference in average values at sites distant and proximal to the outfall removes the coherent temporal oscillations that occur on a seasonal and interannual basis. In effect, parallelism assumes that the large temporal excursions in indices occur in unison at all stations and represent a simultaneous response to regional influences. Time series of differences in parameters at ZID and distant stations are presented in Figure 4.17 on Page 4-49.

If the MBCSD discharge were negatively impacting the benthic community over time, the infaunal community indices at stations near the outfall (B4 and B5) would slowly diverge from those of the more distant stations (B2 and B7). This would appear as a trend or slope in the difference of average indices shown in Figure 4.17. A regression test for a statistically significant trend (slope) in the difference is equivalent to a test of the null hypothesis that the population responses at impacted and non-impacted sites are parallel over time.

Parallelism tests applied to the 31-year time series found slight, but statistically significant trends in all the indices except the *ITI* ($p=0.28$) and dominance ($p=0.06$). The significant, positive linear trends in density, number of species, diversity, and richness demonstrate that the time histories at the ZID and reference stations were not parallel. Instead, they demonstrate that these indices have been steadily increasing at the ZID stations, as compared to the reference stations. Similarly, dominance was found to be decreasing at stations close to the outfall, as compared to distant stations, although its trend was not statistically significant. All of these temporal trends are opposite of those that would occur if the MBCSD the effluent discharge was having a deleterious impact on the benthos near the outfall. Increasingly greater abundance, diversity, and richness, and declining dominance at stations close to the outfall reflects an increasingly healthier infaunal community compared to the distant reference stations.



Because the observed trends are opposite of what would be expected from a community that was being negatively impacted by the MBCSD discharge, the parallelism tests are irrelevant to the impact assessment even though parallelism was rejected for four of the six indices. Accordingly, their p -values are enclosed in brackets in Figure 4.17 to reflect their inapplicability. This was the case for all of the infaunal community indices except the nearly imperceptible negative trend in the ITI , which was so small that parallelism was strongly indicated ($p=0.28$).

As with the spatial gradients in the infaunal indices analyzed for the 2016 survey in the previous section, the reasons for a potential spatiotemporal trend related to outfall proximity is unclear. However, in contrast to the analyses in the previous section, many of the much-more robust parallelism tests presented in Figure 4.17 have a high degree of confidence. The test on numbers of species, for example, indicates that a spatiotemporal trend of the observed magnitude would occur by chance only one-in-a-thousand times. Even so, causality cannot be easily ascribed to an observed relationship where infaunal health is steadily improving in samples collected close to outfall. In fact, it may be related to long-term zoogeographic changes throughout northern Estero Bay as a whole.

4.3.5 Infaunal Community Structure

Determining whether adverse biological conditions exist near the MBCSD outfall involves assessing whether the existing benthic environment supports a balanced indigenous population (BIP). A BIP is an ecological community that exhibits characteristics similar to those of nearby, healthy communities existing under comparable, but unpolluted, environmental conditions. In evaluating a BIP, infauna have an important advantage over other marine organisms, such as marine mammals or finfish. Infauna are relatively easy to collect in numbers large enough for reliable statistical testing, and they do not move about over large areas, so their response is site specific.

In addition, decades of analysis has demonstrated that benthic pollution affects marine ecosystems by changing the infaunal community structure in well-defined ways, namely, changes in the number and type of benthic organisms found within the sediments. However, evaluating the well-being of the infaunal community, and testing its variation over time and space is complicated by the multitude of individual taxa collected within each replicate grab sample. Subtle changes in community composition are not always readily apparent in the large volume of raw data generated by field surveys.

The multivariate complexity of infaunal abundance data makes a BIP analysis challenging because it is difficult to summarize the health of the community into a concise parameter that is indicative of existing environmental conditions. Biodiversity is a common indicator of the well-being of ecological systems, and forms the cornerstone of most impact-assessment studies. It was the basis for the impact assessment described in the previous two sections. However, as described in Section 4.1.4 and Appendix B.1, diversity and other univariate parameters can miss subtle changes in community structure that are reflected by differences in the abundance of individual taxa, but not in the individual indices. For example, two grab samples can have identical infaunal indices, without having a single species in common. One sample may consist entirely of opportunistic pollution-tolerant taxa, and the other sample could contain high numbers of pollution-sensitive taxa, but that important distinction would be completely missed by an analysis of the infaunal community indices alone.

Despite the univariate indices' inability to capture potential changes in community structure, most benthic monitoring studies routinely report standard infaunal community indices, and rely on them in their examinations of temporal and spatial trends. As described in previous sections, the historical record of univariate analyses provides a useful backdrop for evaluating the large infaunal dataset, but the potential for changes at the species level must also be examined. One of the difficulties with infaunal indices is that they only reflect the diversity within a local area (i.e. within an individual replicate grab sample) at a particular time (α -diversity). In contrast, pollution and other stresses tend to induce changes in community composition wherein completely different (pollution-tolerant) taxa begin to inhabit an impacted area. This results in infaunal assemblages that differ substantially across the samples collected through time at widely separated locations. This kind of diversity is measured by β -diversity (Smith et al. 1979). Thus, analysis of individual measures of α -diversity within samples is not well suited to an infaunal dataset containing moderate levels of β -diversity (heterogeneity), such as in the 2016 infaunal dataset. Compositional differences among the 35 samples collected during 2016 were moderately large, as demonstrated by a Whittaker (1972) β -diversity of 2.0.¹ This high level of β -diversity suggests that the summary univariate indices may be incapable of differentiating subtle trends in community structure among the samples.

Multivariate techniques are much better suited to the evaluation of datasets with high β -diversity. The 2016 dataset used in this multivariate analysis consisted of a matrix of counts for 60 taxa² in 35 samples. Analyzing this complex matrix for potential differences among samples by comparing differences in each individual species is intractable, not just because of the large numbers of species, but because many pairs of samples have few species in common. In response to these challenges, landscape ecologists have, over the last 70 years, developed specialized multivariate analysis tools. These techniques are highly effective at extracting the dominant patterns from complex species-sample databases. The multivariate techniques

¹ β_w measures the ratio of the total number of species in the dataset to the average number species in the samples, minus one. If $\beta_w=0$, then all the samples have the same number of species. $\beta_w=2.0$ indicates that the entire 2016 database had three distinct communities, with three-times as many species (43) as the average individual grab sample (14.4).

² Because the multivariate analysis examines differences in community structure among the samples, and not specifically diversity, additional specimens could be included in the database. These additional specimens were not identified to species level, usually because they were juveniles and too small for reliable taxonomic identification.

employed here exploit redundancies (correlations) in the abundance of individual species among the samples, and compress the data into the most meaningful patterns.

Inherent Sampling Uncertainty

As with the other field measurements, it is crucial to determine the inherent uncertainty associated with characterizing an infaunal community at a given station. Specifically, the variability among repeated replicate grab samples collected from an individual station at a particular time lends insight into the confidence in average values reported for each station. Thus, accurate determination of this within-station variability is essential for testing hypotheses concerning the significance of perceived differences between samples collected at different locations. If the inherent sampling variability is statistically larger than the observed difference between the two stations, then the spatial difference cannot be considered statistically significant. As described below, the multivariate analyses of spatial trends in the 2016 database demonstrate that the infaunal communities within most stations cannot be readily distinguished from that of adjacent stations, although the communities between groups of widely separated stations are perceptibly different.

A dendrogram (tree diagram) produced from the multivariate taxonomic database provides a convenient graphical means for determining the relative magnitudes of within-station and between-station infaunal variability among the samples (Figure 4.18 on Page 4-52). Testing the relative variability among the 35 individual grab samples collected during 2016 indicates whether differences among replicates collected at a station are smaller than the differences between the stations, which would indicate that spatial differences are large enough to be reliably resolved, even in the presence of inherent sampling uncertainty associated with defining the station averages.

The relative similarity in the infaunal assemblages among samples, or groups of samples, is graphically displayed using a dendrogram, a branching (tree) diagram. Sample pairs that have the most similar assemblages coalesce into a branch toward the left on the percent-information-remaining axis. The horizontal axis at the top of the dendrogram reflects the amount of information (variability) that is left in the sample-species matrix after two samples are combined. Combining samples with relatively similar species composition results in a minimal loss of information (variability) in the original species-sample matrix. For example, the short, almost non-existent branches connecting replicate sample pair at the top of the southern group¹ in Figure 4.18 shows that the infaunal assemblages within those samples were nearly identical, and considering them as a single combined assemblage results in a negligible (<0.1%) loss of multivariate information. As more samples, or other groups, are added to each branch, the differences in the assemblages associated with each group become increasingly larger, and as each group is combined into a new group, more overall information about the differences in the assemblages is lost.

A multivariate analysis of variance (MANOVA) applied to the dendrogram quantifies the clustering tendency as a whole. It determined that the within-station similarity among replicate samples was far greater than expected by chance ($p < 0.0002$). As with the hypothesis tests throughout this report, the p -value cited here, and throughout this section, is the probability that the observed station-clustering tendency could have occurred by chance alone. It is determined by comparing the magnitude of the difference between stations, with the scatter in the replicate samples collected at a particular station. If the scatter in replicate samples is large compared to the difference in sites, then the station difference is more likely to be an artifact of chance alone, rather than a real spatial difference.

¹ The branch connecting the fuchsia diamond associated with Replicate 4 of Station B4 (B4R4) and the red square associated with Replicate 5 of Station B5 (B5R5)

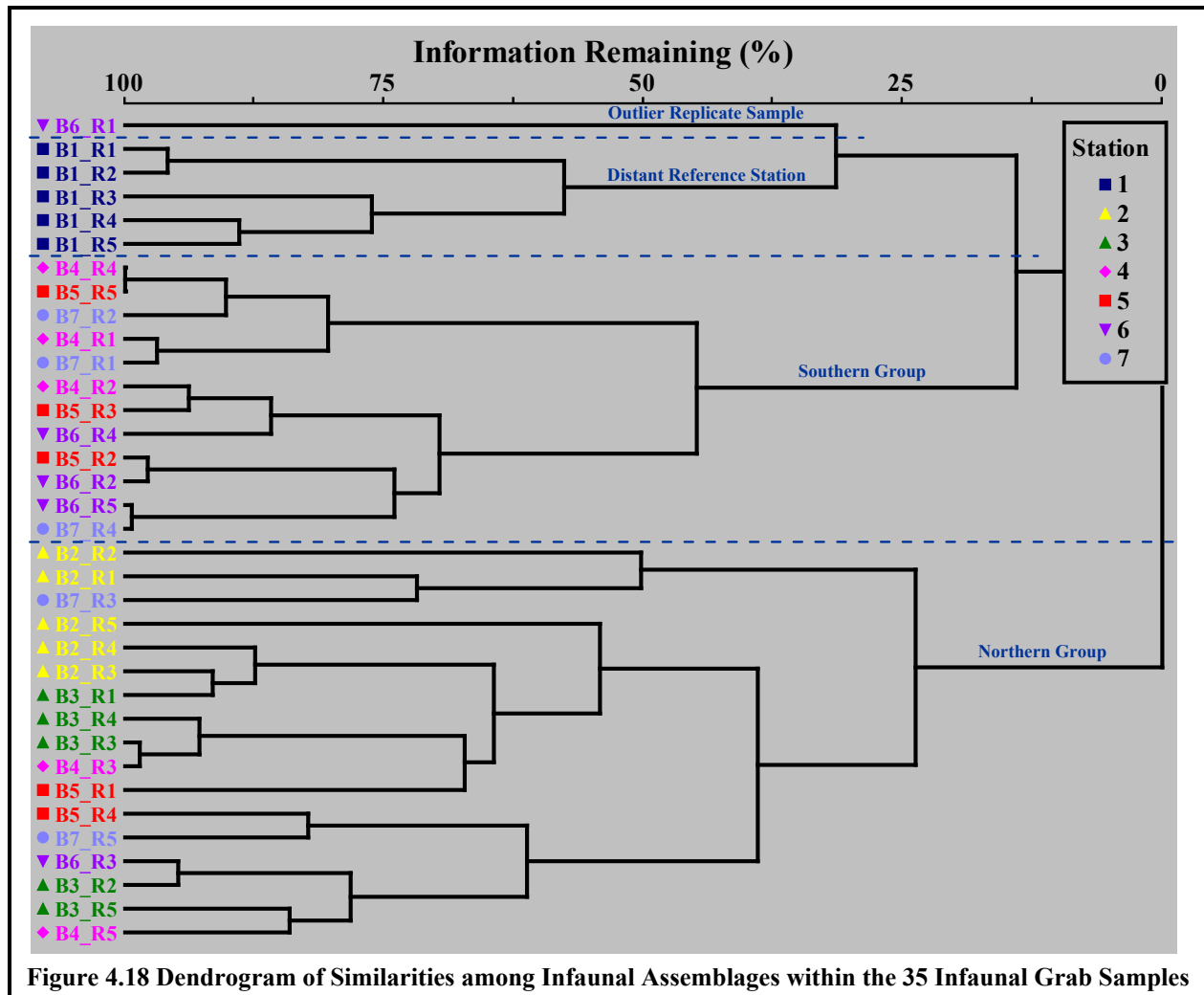


Figure 4.18 Dendrogram of Similarities among Infaunal Assemblages within the 35 Infaunal Grab Samples

Generally, differences in groups are considered statistically significant when p -values are less than 0.05, which corresponds to a 5% (1-in-20) probability that the difference would have occurred by chance alone. Thus, the observed infaunal differences among the seven stations during 2016 were very unlikely (0.02% chance) to have been entirely due to random sampling uncertainty. Although, the 60-by-35 species-sample matrix contains discernible information about spatial differences in the infaunal communities, this does not mean that the communities within each pair of stations were perfectly distinct from one another. Instead, it only implies that the within-station fidelity among all 35 replicate samples was definitely greater than if the replicate samples were randomly assigned to stations.

In fact, only Station B1 supported an infaunal community that was sufficiently unique to produce replicate samples with perfect within-station fidelity. The “*Distant Reference Station*” branch in the upper portion of the dendrogram in Figure 4.18 has all five of its replicates clustering first with other samples from Station B1. The likelihood of this happening by chance is low ($p < 0.027$) and demonstrates that the infaunal community at this location is fundamentally different from that which exists within the survey area surrounding the outfall 1 km to the south.

This finding is consistent with past surveys, namely zoogeographic differences over the large distance that separates Station B1 accounts for the unique infaunal community that resides there, and for other differences in the benthic environment there that are described in Section 4.2. In particular, Station B1's more protected location in northern Estero Bay results in its consistently higher mud fraction (black bar in the lower right bar chart of Figure 4.6). Mud fraction closely correlates with the infaunal assemblage present within a given sediment sample. Additionally, Station B1 has had consistently higher sand dollar densities over the past seven-year post-recruitment period, including the 2016 survey (black squares in Figure 4.12). Because Station B1's different benthic habitat confounds the evaluation of conditions within the survey area immediately surrounding the outfall, the original requirement for sampling at that station was removed when the current discharge permit was issued. Although it continues to be sampled for historical consistency, the inclusion of data from Station B1 skews the multivariate analyses substantially, masking subtler differences in the community structure among the remaining stations far to the south.

In contrast to the survey-wide MANOVA, one conducted on individual pairs of stations demonstrates that many pairs of stations within the survey area immediately surrounding the outfall contained communities that were not easily distinguishable from one another. Only five of the fifteen station-pair comparisons contained replicates that reliably differentiated their communities ($p < 0.016$). All were associated with pair-wise comparisons between the Upcoast Nearfield and Midfield Stations B2 and B3 with the four stations to the south B4, B5, B6 and B7. This finding suggests that stations located near each other had infaunal communities statistically indistinguishable from one another, but that the groups of stations located on the opposite ends of the survey area contained dissimilar sets of infauna, albeit only marginally.

The low station-fidelity among replicate samples from individual stations within the survey area is reflected by the left-most dendrogram branches in Figure 4.18; not a single replicate sample from a given station clustered first with another replicate from that same station. The subtle north-south infaunal gradient only becomes apparent after many replicate samples have been combined and only 23.8% of the information (variability) in the species-sample matrix remained. At that point, all of the replicates from the two upcoast Stations B2 and B3 have coalesced within the "*Northern Group*." Although this group also includes several replicates from the other stations, a many replicates from ("*Southern Group*") Stations B5, B6, and B7 contained an infaunal assemblage that was distinct from that of Stations B2 and B3.

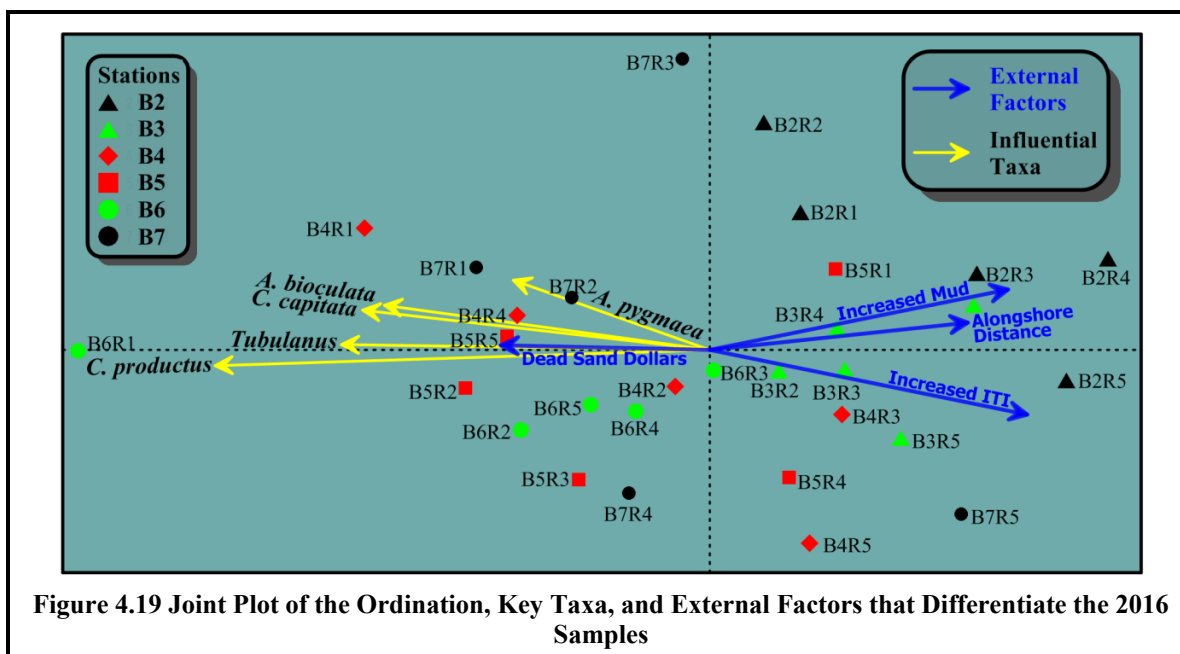
The most striking example of the wide infaunal variability among replicate samples collected during 2016 is the prominent outlier found within the first replicate collected at Station B6 (B6R1) at the top of Figure 4.18). The taxa contained within that sample departed significantly from all of the other 34 samples collected during 2016. Although it eventually coalesced first with the samples from Station B1, the similarity was very marginal, and arose only when less than 32% of the information remained in the database. In contrast to the infaunal community found within all the other samples collected during the 2016 survey, its assemblage exhibited many characteristics of an organically degraded benthos similar to the sand-dollar dominated community found throughout the region during the 2015 survey (MRS 2016a). For example, the prevalence of subsurface detritus feeders resulted in an *ITI* (35 in Appendix Table B.9) that was far lower than that of other 2016 samples, but comparable to many of the 2015 samples. As described in Section 4.3.3, it is not surprising that a potentially significant lower average diversity and higher dominance was found at Station B6 (light green bar in Figure 4.16ab). It is likely that the difference in the average indices determined that station was solely due to the highly unusual infaunal community that was found only in the first replicate grab sample collected at that station.

Spatial Variability

The two major branches in the dendrogram of Figure 4.18 demonstrate that most replicate samples collected at stations south of the diffuser structure (Stations B5, B6, and B7 in the upper branch), contained an infaunal community that differed from that of the group of stations upcoast of the diffuser structure (Stations B2, B3, and B4 in the lower branch). This north-south trend suggests a naturally occurring spatial pattern of along-shore change rather than a pattern indicative of impairment surrounding the outfall, which would be reflected by groupings related to outfall proximity. However, the influence of external influences on community composition can be better explored using another related multivariate analysis technique, known as ordination analysis.

As described above, zoogeographic differences over the large distance that separates Station B1 have historically accounted for the unique infaunal community that resides there. This was also the case during 2016 because of its consistently greater sand-dollar density (Figure 4.12) and greater mud fraction (Figure 4.6). Because of these differences, the ordination analysis focused only on the subtler differences in the community structure among the stations immediately surrounding the outfall.

The resulting two-dimensional ordination¹ diagram provides a visual interpretation of the differences in the infaunal communities found among the 30 samples remaining after exclusion of Station-B1 data (individual color-coded sample points in Figure 4.19). The diagram distills the major infaunal differences among samples as a separation between sample points in the multivariate hyperspace. As such, the diagram's axes do not represent distances in physical dimensions. Instead, the locations of the points within the diagram characterize the composition of the assemblage within individual benthic grab samples. The distance between a pair of points measures the degree of difference in the species compositions contained within those samples. Widely separated points (e.g., the green circle of B6R1 at



¹ The multivariate ordination was derived from a non-metric multidimensional scaling (Mather 1976, Kruskal, 1964) of Sørensen (Bray-Curtis) distance (Sørensen, 1948), the same multivariate distance measure used in the dendrogram. There was a 0.4 percent chance ($p=0.004$) that the stress measure of the final solution's fit could have occurred by chance based on Monte Carlo randomization tests. The method is unconstrained by the external factors, but the influence of those factors was evaluated by superimposing them on the ordination diagram.

the extreme end of the negative X-axis and the black diamonds of B2R4 and B2R5 on the opposite extreme of the X-axis) have dissimilar infaunal assemblages, namely, few species in common, and large differences in the abundance of those species that are common between the samples. In contrast, points that lie close to one another have assemblages with many common species and similar abundances among those common species (e.g., three of the other green circles associated with replicate samples R2, R4, and R5 at Station B5 located closer to the origin in the lower left quadrant).

In addition, separation between sample points along the horizontal axis of the ordination plot characterizes a variation in species composition that fundamentally departs from the variation in community composition along the vertical axis. In past surveys, changes along the major (horizontal) axis often reflected a large-scale zoogeographic difference while the secondary vertical axis usually coincided with a local environmental gradient in the samples, such as water depth or mud content. However, the vertical distribution of sample points within the 2016 ordination diagram did not exhibit a strong association with external influences that were commonly observed in past surveys.

Mud Content and Alongshore Location

The diagram (Figure 4.19) resulting from an ordination of the 2016 database echoes some of the findings discussed above in conjunction with the dendrogram (Figure 4.18). Specifically, replicate samples from a particular station did not contain taxa that were most similar to those of other replicates from that same station. Instead, data points associated with replicates from each station tend to be widely dispersed across the ordination diagram of Figure 4.19, rather than in close proximity to one another. Additionally, the outlier replicate from Station B6 was well-separated from the cloud of other data points (green circle associated with B6R1 on the far left), indicating that it contained an infaunal community that was fundamentally different from all the other samples collected during the 2016 survey.

Lastly, the only perceivable pattern in the sample-point distribution was that upcoast Stations B2 and B3 (green and black triangles) were all located along the positive X-axis; while most replicates from the southernmost Stations B6 and B7 (green and black circles) were located along the negative X-axis. More importantly, a pattern indicative of changes in infaunal composition related to outfall proximity was not apparent, indicating the discharge was not perceptibly influencing the infaunal community surrounding the outfall. Impacts to the infaunal community from the MBCSD discharge would result in an ordination diagram where the ZID stations (in red) form a cluster that is distinct from that of the near-field stations (green), and from that of the mid-field and reference stations (black). No such consistent grouping or steady progression of colors is apparent in the ordination diagram.

Another aspect of the ordination analysis provides a rigorous method for evaluating the importance of various external environmental factors in determining the infaunal community within individual replicate samples. The most influential of these factors are jointly displayed on the ordination diagram of Figure 4.19 as blue arrows radiating from the center of the diagram. The angle and length the arrows reflect the direction and strength of the relationship between each external factor and the infaunal community within individual samples displayed as points in the ordination diagram. Other external factors were also examined but did not exhibit a significant relationship to the sample distribution, and consequently, the arrows were much shorter than those shown in the ordination joint plot. Any influence of outfall proximity on the sample-point distribution, for example, was far too weak to be of significance ($p=0.28$), while alongshore location was highly significant ($p<0.001$).

As indicated by the “*Alongshore Distance*” arrow, the infaunal community within samples collected in the northernmost portion of the survey area during 2016 differed perceptibly from that found in most samples collected well to the south (on the left side of the ordination diagram). However, the actual external environmental factor(s) responsible for the spatial difference in communities is equivocal.

Prevailing oceanographic conditions may be more quiescent at the northernmost stations, but the specific aspect of the benthic environment that caused the infaunal differences is unclear. Decreased wave and bottom-current energy can also change the sand-dollar distribution as well as the mud content of surficial sediments. Both have a profound impact on infauna. Within the 2016 database, increasing mud content was more closely associated with the distribution along the positive X-axis than location alongshore. Its greater strength of correlation is reflected by the longer “*Increased Mud*” arrow.

Feeding Guilds and Dead Sand Dollars

In contrast to some prior surveys, the distribution of 2016 samples along the Y-axis of the ordination plot was not readily ascribable to an external factor that was unrelated to the influential factors that controlled the sample distribution along the horizontal axis. Instead, the trophic health of the infaunal community, as measured by the *ITI*, tended to increase with northward location within the survey area, and with the associated increase in mud content. However, the increase in clean-sediment feeding guilds was not closely correlated with mud content and location. Although the “*Increased ITP*” arrow pointing into the lower right quadrant of Figure 4.19 was as long as the “*Increased Mud*” arrow, indicating it had an equally strong influence on sample-point distribution, it was not perfectly aligned with the other two arrows.

Nevertheless, the strength of the *ITI* relationship indicates that the locations of many of the sample points within the ordination diagram were related to the feeding guilds of the infauna within those samples. For example, samples located on the left half of the diagram and opposite of the “*Increased ITP*” arrow tended to have reduced *ITI* values, indicating an increased presence of subsurface detritus feeders in sediments richer in organic material. Because the sample-point distribution in the ordination diagram was not related to outfall proximity, wastewater discharge from the MBCSD outfall is not a plausible source for the increase in sediment organics. Moreover, the *ITI* reductions did not appear to have any spatial coherency whatsoever; sharply reduced *ITIs* were measured in an isolated replicate sample from a given station, while all the other replicate grab samples collected at that same station had normal *ITIs*. A widespread source of organic contamination would be expected to produce a more *ITI* consistent response within all the replicate samples from a given station.

Determining the nature of the increased organics within isolated replicate samples was initially perplexing because the concentrations of organic constituents are only measured within a composite of three chemistry grab samples at each station. As a result, there was no direct information about the organic content within a particular infaunal grab sample with low *ITI*. Moreover, because composite samples were analyzed for organic content at each station, it is not surprising that the organic loading results did not exhibit large outliers (Figure 4.6). It is noteworthy however, that volatile-solids concentrations measured in the last three surveys have been consistently higher than any measured in the prior 28 years (Figure 4.9c). Numerous large sand dollars also became prevalent during that same period.

The very large, sample-specific *ITI* decline within outlier Sample B6R1 ultimately provided the needed insight into the source of the organics. That extreme outlier contained a community so different from all the other samples collected during the 2016 survey, that the source of a markedly increased organic content must be readily apparent to have caused such a profound impact. The vastly different community within that replicate is reflected by its isolated location on the far left of the ordination diagram, and well away from all the other samples. In addition, because its location was directly opposed to the “*Increased ITP*” arrow, its unique community was dominated by detritus feeders and a low *ITI*. At 35, the *ITI* measured within replicate grab sample B6R1 was the lowest measured among the 35 samples collected during 2016 by far (refer to the farthest right column of Table B.9 in Appendix B). As described previously, *ITI* values that low were more typical of samples collected during 2015 when infauna

consisted almost entirely of detritus feeders in the presence of numerous large sand dollars that dominated the benthic environment both in their physical presence, and in their taxonomic influence.

Because of their important role in recent surveys, the 2016 sand-dollar enumerations were re-evaluated in an attempt to identify the cause of the marked increase in detritus feeders within specific replicate samples, particularly the outlier sample. Upon retrieval of each infaunal grab sample offshore, sand dollars were washed and removed from the sieve and stored separately. Subsequently, the sand dollars were enumerated onshore. As with all specimens, live adult and juvenile organisms were separately enumerated, while fragments¹ and dead² specimens were counted but not included in the database.

As part of the ordination analysis, the influence of sand-dollar density is evaluated by including it as an external factor. In many past surveys, sand dollar abundance was found to be an influential factor in distinguishing the samples within the ordination diagram, and its external-factor arrow was plotted along with those of other influences. However, this was not the case with the 2016 infaunal database. Its lack-of-influence was initially puzzling until it was realized that the (live) sand-dollar densities excluded the potentially important component of dead specimens. Deceased sand dollars were rarely encountered in prior surveys and were therefore, not considered important.

The subsequent addition of another external factor based on the abundance of dead sand dollars within individual replicates collected during 2016 was telling (“*Dead Sand Dollars*” arrow along the negative X-axis). Specifically, over half of the samples collected within the survey area contained less than three dead sand dollars (Figure 4.20a), and their sample points were largely located in the right hemisphere of Figure 4.19. The remaining samples contained between three and nine dead sand dollars and were scattered throughout the left hemisphere of Figure 4.19, except for the outlier Replicate B6R1, where over half (26) of the sand dollars were dead (the right side of Figure 4.20b).

In recent prior surveys, the presence of numerous large sand dollars dictated the character of the infaunal community within the survey area. They reduced sediment-sample volumes because of their physical size, and exerted their taxonomic influence because of competition resulting from a change in their feeding mode. However, during 2016, the senescence of that 2009 cohort began to introduce decaying organic material within highly localized

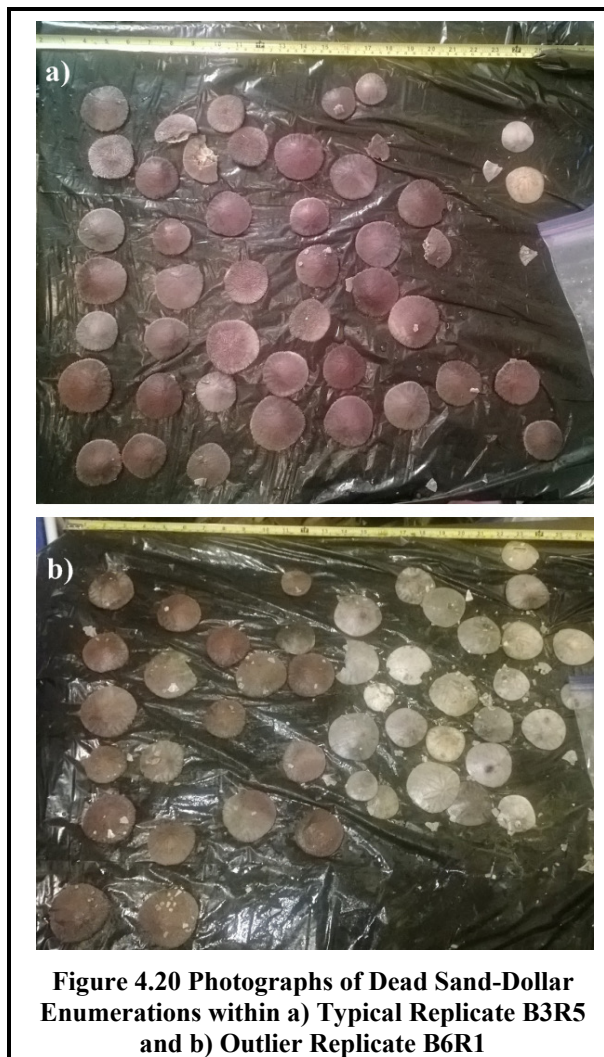


Figure 4.20 Photographs of Dead Sand-Dollar Enumerations within a) Typical Replicate B3R5 and b) Outlier Replicate B6R1

¹ Pieces of sand dollars that do not encompass half of a disk are considered fragments to avoid double-counting pieces from a single specimen.

² Organisms that were dead when they were collected offshore

areas. That organic load dramatically altered the character of the resident infaunal community in ways unlike the influence of live sand dollars in prior surveys.

Sand dollar senescence is rarely observed in the field because their populations are normally controlled through predation by ochre sea stars (*Pisaster ochraceus*) and other related genera. This species is considered a classic keystone species whose wellbeing profoundly affects intertidal regions. Because of its sand-dollar predation in subtidal regions, the natural longevity of sand dollars has not been well established in prior studies. However, the onset of the sea-star wasting syndrome in recent years has largely eliminated this prominent sand-dollar predator. As a result, the field data collected during the 2016 MBCSD survey lent unusual new insight into sand dollar longevity and sand-dollar life history. The new insights also foretell of major future changes to the benthic environment within the MBCSD benthic survey area.

First, the MBCSD survey data delineated the minimum natural longevity of sand dollars at seven years, which is the time between recruitment of the 2009 cohort, and the initial onset of senescence observed within the 2016 MBCSD database. Second, a massive sand-dollar die off can be expected after 2006 because of their unchecked senescence. Furthermore, based on the subtle zoogeographic trend observed within the survey area during the 2016 survey, this die-off has already begun in the southern portion of the survey area, and is likely to extend to the north. Accordingly, a few months after the October 2016 survey, beaches along northern Estero Bay were inundated with sand-dollar tests that had washed ashore near the location of the survey area (Figure 4.21).

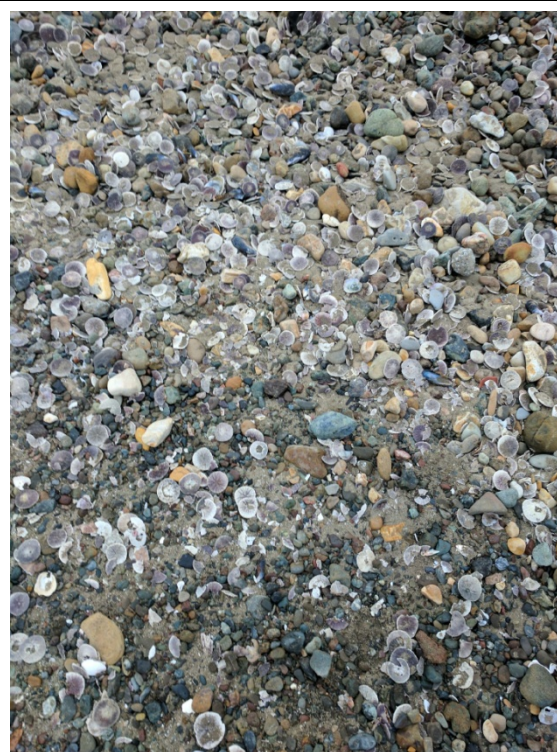


Figure 4.21 Photograph from 2 February 2017 of numerous sand-dollar tests along the beach adjacent to the MBCSD survey area¹

Lastly, a fundamental change to an infaunal community of detritus feeders living in organically enriched sediments can be expected to occur within the MBCSD benthic survey area in future years. This trophic shift will be of unknown duration as senescence of the 2009 cohort progresses. However, the resulting shift in key taxa, some of which are described below, could be incorrectly ascribed to organic loading from the MBCSD discharge if the naturally occurring senescence is not considered during analysis of future benthic surveys.

Key Taxa

The abundances of individual species within a given replicate sample, as compared to that of another sample, are difficult to discern and interpret when an investigator is faced with the enormous amount of information contained in the 2016 infaunal database, as listed in Table B.1. Although perusal of the table may reveal patterns for specific taxa, it is difficult to decide how important the differences in counts of

¹ Photograph provided by Bruce Keogh, former Wastewater Division Manager of the MBCSD WWTP

one taxon are, relative to another. Additionally, many species co-occur and yield the same distribution among samples, further confusing the interpretation. Finally, the most abundant taxa are often not the most diagnostic of potential impacts because they are present in substantial numbers within all the samples. As described previously, multivariate community analysis was designed to distill this complex set of information quantitatively, to eliminate redundant or marginal patterns, and to produce most important information concerning how the communities within individual samples relate to one another. In that regard, the ordination methodology also identifies which taxa primarily influence the community differences among replicate samples. Those influential taxa often have life-history information, such as a tolerance or sensitivity to pollution and organic loading, which can lend further insight into the potential impacts from effluent discharge.

An indicator species analysis compliments the multivariate sample analysis described above, and is graphically overlain in yellow on the joint plot of Figure 4.19. As with the external environmental factors, the level of influence each individual key taxon had on the overall distribution of the samples is represented by the yellow arrows (vectors) superimposed on the ordination plot. The length of an arrow for an individual taxon reflects how strongly it participated in differentiating the samples, and its direction points toward samples that have a comparatively elevated abundance of that taxon. Conversely, arrows pointing away from samples reflect a comparatively low abundance of an indicator taxon.

Because the infaunal community within Sample B6R1 was so decidedly anomalous, its dominate taxa influenced the distribution of sample points in the ordination diagram to the greatest degree. All five of the key taxa exhibited much higher abundances within that sample as compared to most of the 34 other samples collected during the 2016 survey.

Two of these key taxa, the polychaete worms *Capitella capitata* (Figure 4.22) and *Armandia bioculata* (Figure 4.23), are opportunistic detritus-feeding organisms. Their presence is considered diagnostic of organically polluted sediments because they are capable of rapid population growth in highly disturbed habitats. As such, their increased presence in Sample B6R1 contributed to the low *ITI* determined for that sample.

However, because only three *C. capitata* specimens were found in that sample, they only contributed minimally to the reduced *ITI* in that sample. Nevertheless, their presence in the 2016 samples was highly unusual because they have not been encountered in any of the MBCSD benthic samples collected since 2010. The presence of three more specimens in Sample B7R2 and two in B7R1 (two black circles in the upper left quadrant of Figure 4.19) during the 2016 survey was even more unusual, and suggests a source of decaying (sand-dollar) organic material may have been present in those samples as well. Regardless, because of their location on the southernmost extent of the survey area, the source of the organic material is not likely to have been effluent discharge.

In contrast to the relatively minor *ITI* contribution from *C. capitata* specimens, the 43 specimens of *A. bioculata* present in the outlier sample B6R1 was almost entirely responsible for the anomalously low *ITI* determined for that sample. Furthermore, all the specimens were juveniles, suggesting a localized and recent colonization.



Figure 4.22 *C. capitata*

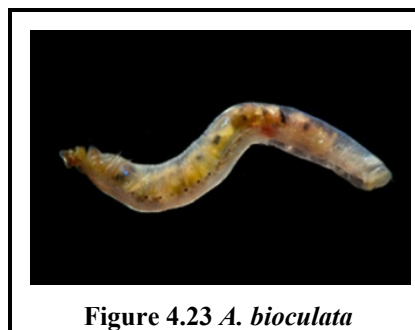


Figure 4.23 *A. bioculata*

Nonetheless, unusually large numbers of *A. bioculata* specimens were also present in Samples B4R1 (17 counts) and B7R1 (11 counts), which further explains their location along the negative extreme of the X-axis. This Opheliid lugworm is a non-selective subsurface detritus feeder that normally thrives in sediments that are organically enriched below the sediment interface. However, *A. bioculata* is also known to thrive within sand dollar beds (Smith 1981). Field studies have consistently found significantly higher numbers of *A. bioculata* within sand dollar beds than within adjacent sediments with identical concentrations of organic material. Thus, the large *ITI* reduction observed within this particular sample may not be solely related to degradation in sediment quality, but instead the lugworm's natural affinity for sand dollar beds may have contributed to the observed *ITI*.

Another key opportunistic annelid worm, *Apoprionospio pygmaea*, tends to co-occur with *A. bioculata* within sand dollar beds. Accordingly, *A. pygmaea* was also more populous in the outlier sample B6R1 (26 counts) than in the two closely related Samples B4R1 (19 counts) and B7R1 (16 counts). Because they are members of the Group II surface detritus-feeding guild, their increased presence in the samples located along the negative X-axis also contributed to the reduced *ITIs* among samples collected in the southern portion of the survey area, although their influence on the computation is far smaller than that of Group IV taxa.

Two other key taxa were noteworthy, not only because they covaried with the other indicator species described above, but because they have not been regularly observed in past surveys. The nemertean tube-dwelling worm *Tubulanus* spp. is not part of the feeding guilds incorporated in the *ITI*, and thus the three specimens found in the outlier sample B6R1 did not contribute to its low *ITI*. Nevertheless, it was consistently found in conjunction with the other key taxa and thus, has some affinity to organic material associated with dead sand dollars.

The last key taxon, *Cancer productus*, is a red rock crab that has never been observed within the MBCSD survey area, although it is prevalent throughout the central coast region. All of the specimens found in the 2016 MBCSD samples were juveniles, but could be reliably distinguished from *Cancer gracilis* that commonly occurs in the database. The distribution of *C. productus* specimens was again noteworthy, because five of the seven specimens of this new taxon were found in the outlier sample B6R1.

Overall, the multivariate ordination demonstrates that the differences in the infaunal community structure at various sampling locations were largely controlled by the presence of organic material from dead sand dollars. Additionally, alongshore location, mud fraction, and feeding guilds were also covarying influential factors. Overall, the multivariate results were consistent with the findings from the analyses of infaunal community indices and sediment chemistry. Together, the weight of quantitative evidence demonstrates that, even within sediments immediately adjacent to the outfall structure, there was no evidence of a buildup of organic wastewater particulates from the discharge of the MBCSD effluent, and that a balanced indigenous population of marine organisms resides throughout the region, including within the area near the MBCSD outfall.

5.0 CONCLUSIONS AND RECOMMENDATIONS

The MBCSD monitoring program was designed to evaluate the performance of the wastewater treatment plant and to monitor the quality of effluent discharged to the ocean. This annual report evaluated compliance with the NPDES discharge permit and documented the potential for discharge impacts through quantitative analyses of an extensive data set of effluent constituents, receiving-water measurements, sediment chemistry concentrations, and marine biological enumerations. Of the thousands of measurements collected as part of the monitoring program during 2016, only one exceeded a discharge limitation and then, only by a very marginal amount.

As discussed in Section 2.2.2, the 106-mg/L TSS concentration measured in an effluent sample on November 4 slightly exceeded the 105-mg/L permitted maximum-daily concentration. The exceedance culminated a slow two-month-long decline in secondary treatment performance that was related to a change in the odor-control compound used in the collection system. A new chemical formulation adversely affected settling within the Plant's treatment process, which was further compounded by all-time low flow through the plant. The corrective action taken was to return to the use of ferrous chloride for odor control, after which effluent TSS concentrations quickly returned to normal. Although the TSS exceedance was slight, its occurrence was highly unusual in a perfect 14-year record of compliance with the solids limitations in the discharge permit.

5.1 TREATMENT PLANT

The MBCSD Wastewater Treatment Plant (WWTP) has been operating under the provisions of a 301(h)-modified NPDES permit since 1986, with 2016 marking the 31st year of consistently high performance by this treatment facility. In fact, many of the best measures of treatment performance have been achieved during the past decade of operation (Table 5.1). For example, the low average effluent Total Suspended Solids (TSS) concentration during 2016 (37 mg/L) was the direct result of the treatment process's exceptional efficiency, which removed over 90% of the influent solids. As a result, the solids discharge was only 21% of that allowed under the NPDES permit.

Removal of oil and grease (O&G) and biochemical oxygen demanding (BOD) materials from the wastewater stream also attest to the plant's overall high standard of performance during 2016. As with TSS, the discharge of these materials was only a fraction of the allowed and was lower than average emissions over the past three decades of operation. Although the 2016 removal rates were comparable to prior years, the exceptionally low flow during 2016 helped reduce the overall emission of these substances.

5.1.1 Overall Performance

Measurements of wastewater characteristics acquired throughout 2016 demonstrate that the treatment process exceeded expectations based on the original plant design, the NPDES discharge permit, and federal regulatory standards. For example, during 2016, the plant removed 15% more suspended solids than the permitted minimum, and the BOD removal rate was nearly two-and-a-half times greater than the minimum permitted rate of 30%. This was the case even though, during nearly half of the year, effluent BOD concentrations remained at or below the 60-mg/L threshold where minimum 75% removal-rate requirements are considered applicable (See the recommendation in Section 5.4). Similarly, effluent TSS concentrations remained far below that concentration threshold throughout the entire year.

Table 5.1 Average Annual Wastewater Parameters

Year	Flow (MGD)	Suspended Solids				Biochemical Oxygen Demand			
		Influent (mg/L)	Effluent (mg/L)	Removal (percent)	Emission (MT)	Influent (mg/L)	Effluent (mg/L)	Removal (percent)	Emission (MT)
1986	1.42	332	32.8	89.8	64	235	77.0	67.2	151
1987	1.51	274	21.8	92.0	45	257	52.0	79.8	108
1988	1.51	397	29.8	90.0	62	242	43.9	81.9	92
1989	1.46	321	37.3	88.4	75	259	69.8	73.1	141
1990	1.38	345	36.0	89.6	69	261	75.7	71.0	144
1991	1.28	280	30.5	89.1	54	236	66.9	71.6	118
1992	1.41	310	43.0	86.3	84	224	59.3	73.5	116
1993	1.54	339	33.0	89.6	70	222	39.0	81.9	83
1994	1.38	310	32.0	89.4	61	249	33.0	86.4	63
1995	1.55	270	30.6	87.6	69	208	31.4	83.9	67
1996	1.55	344	33.1	89.9	70	241	35.7	85.0	73
1997	1.64	283	36.0	86.6	79	231	38.6	83.0	85
1998	1.95	236	38.8	83.9	101	216	39.1	81.5	99
1999	1.68	386	44.0	86.7	102	287	49.5	82.5	118
2000	1.77	337	37.4	87.5	91	271	50.3	81.1	125
2001	1.48	450	37.6	89.5	74	396	62.7	83.1	127
2002	1.14	374	49.2	86.0	77	386	67.5	82.4	101
2003	1.06	314	39.2	86.7	56	311	56.3	81.3	81
2004	1.09	354	28.9	91.3	44	336	53.3	83.8	81
2005	1.25	373	24.3	93.3	42	303	49.8	83.0	88
2006	1.19	335	20.5	93.2	34	291	45.3	83.8	75
2007	1.09	381	20.9	94.1	31	330	44.4	86.0	68
2008	1.10	337	20.0	94.1	30	331	38.4	88.3	58
2009	1.09	328	25.3	92.3	38	311	37.6	87.4	56
2010	1.19	383	26.6	92.1	42	350	49.4	85.2	85
2011	1.24	343	26.8	92.4	45	312	52.2	83.2	89
2012	1.10	379	27.1	92.5	41	322	49.9	83.5	77
2013	0.96	351	29.9	90.4	39	327	55.7	82.6	74
2014	0.94	377	29.2	91.3	37	352	51.4	85.3	66
2015	0.93	389	30.8	91.5	39	370	48.9	86.5	63
2016	0.84	358	37.1	90.0	42	384	63.7	82.9	74
Average	1.31	342	31.9	89.9	58	292	51.2	81.7	92
Limitation	2.06		70.0	75.0	199		120.0	30.0	342

Table 5.1 Average Annual Wastewater Parameters (continued)

Year	Oil and Grease			Mass Emission (MT)	Turbidity (NTU)	pH	Chronic Toxicity (TUC)	Ammonia as NH ₃ -N (mg/L)
	Influent (mg/L)	Effluent (mg/L)	Removal (percent)					
1986	64	13.8	78.4	27	26	7.7		18
1987	44	6.2	85.9	13	23	7.5		
1988	38	6.3	83.4	13	40	7.5		
1989	28	6.1	78.2	12	49	7.4		26
1990	34	8.5	75.0	16	55	7.4		26
1991	73	6.9	90.5	12	50	7.3		18
1992	33	5.3	83.9	10	56	7.3		9
1993	26	6.0	76.9	13	43	7.4	19.42 ¹	20
1994	60	≈4.1	93.2	≈8	36	7.5	4.37	27
1995	63	5.1	91.9	11	32	7.5	4.35	23
1996	52	7.9	84.8	17	34	7.7	4.83	23
1997	49	5.3	89.2	12	32	7.7	7.80	23
1998	51	5.4	89.4	15	34	7.6	7.80	19
1999	52	6.2	88.1	14	48	7.5	5.00	25
2000	74	5.5	92.6	13	39	7.5	5.60	24
2001	47	≈4.6	90.2	≈9	41	7.4	5.60	28
2002	39	≈4.4	88.7	≈7	41	7.5	4.98	31
2003	44	5.3	87.9	7	34	7.5	7.80	27
2004	47	≈3.7	92.0	≈6	26	7.5	5.60	29
2005	62	≈4.4	92.9	≈8	23	7.6	5.60	27
2006	44	≈4.1	90.6	≈7	26	7.6	4.36	28
2007	52	≈4.0	92.4	≈6	27	7.6	4.36	28
2008	84	≈4.4	94.8	≈7	30	7.5	5.56	27
2009	93	≈4.5	95.1	≈7	29	7.5	15.82 ²	32
2010	76	≈4.7	93.8	≈9	31	7.6	10.88 ²	34
2011	75	≈4.0	94.6	≈7	26	7.6	13.95	27
2012	91	5.0	94.5	8	26	7.6	13.95	33
2013	115	≈4.5	96.1	≈6	25	7.5	24.55	40
2014	81	≈4.0	95.1	≈5	28	7.5	17.90	50
2015	81	<1.7	97.9	<2	29	7.5	17.90	45
2016	79	≈3.2	95.9	≈4	39	7.5	17.90	44
Average	60	5.3	89.5	10	35	7.5	9.83	28
Limitation		25.0			75	6-9	134.00	80.4

¹ Screening bioassay of three marine species

² Screening bioassay of two marine species

Moreover, the plant routinely achieved treatment levels that exceeded the removal requirements applicable to full secondary treatment. For example, monthly solids-removal rates substantially surpassed the 85% full secondary criterion in eleven months, while BOD removal rates achieved full secondary levels during three months of the year. Other effluent constituents were also correspondingly low during 2016. The discharge of settleable solids was imperceptible, and monthly averages of effluent turbidity, pH, ammonia, and coliform density remained well within the applicable permit limitations.

The continued high overall performance of the treatment process during 2016 was the direct result of vigilant control by plant personnel, a proactive program of preventative maintenance, and the successful completion of numerous MMRP projects. Plant personnel actively sought out and corrected potential mechanical problems with plant components before they occurred, and responded quickly to the occasional unforeseen failure. During 2016, and in recent prior years, their constant attention to the idiosyncratic disinfection process undoubtedly prevented excursions beyond allowable discharge limits.

The benefits of current and past efforts to reduce rainwater inflow and groundwater infiltration (I&I), and to more accurately measure plant flow, are also evident in the record of annual average flow rates listed in Table 5.1. These efforts account for the marked decline in plant throughput during 2002, when average daily flow dropped below 1.3 MGD and remained at or below 1.25 MGD for the last decade-and-a-half. In all fifteen years prior to that time, annual reported flow remained above 1.25 MGD.

5.1.2 Flow Rate

Historically, the most influential factor affecting flow rate was a metering inaccuracy that resulted in flow overtotalization. A study conducted during 2002 found that rates reported by the effluent flow meter were consistently overestimated by as much as 25%. Without this overtotalization, average flow over the 30-year record would be closer to 1.17 MGD rather than the reported 1.33 MGD. Because of overestimated flow prior to 2002, the annual mass emissions were also overestimated. Consequently, the average emissions over the entire plant history, shown in bold at the bottom of Table 5.1, are slightly inflated. Specifically, over the operational history of the WWTP, the reported average TSS emission of 58 MT was actually closer to 51 MT. Similarly, the average BOD emission reported at 92 MT was, in reality, closer to 81 MT, and the reported 10 MT of O&G discharged was closer to 9 MT. Even after adjusting for past overtotalization in the historical averages, the 2016 emissions of TSS, BOD, and O&G (42, 74, and ≈3.2 MT) were still well below those of the overall operational record (51, 81, and 9 MT).

During 2002, overtotalization was largely eliminated after the more accurate influent flow meter was commissioned, and flow began to be reported based on its measurements. In contrast to the existing turbine flow meter that was located within the effluent-discharge stream from the plant, the new meter determined flow from water level within a metering flume located within the influent stream. However, on occasion, the new meter still overtotalizes flow, for example, when the influent flume becomes temporarily surcharged after water backs up into the influent channel behind the plant headworks. Nevertheless, judicious use of corrected effluent flow totals in place of these obviously aberrant influent flow-meter reports effectively eliminated significant flow overtotalization after 2002. Highly accurate annual flow totals are now regularly achieved by reporting a downward-adjusted effluent-meter flow on those rare occasions that the influent flow data is compromised. The effluent-flow adjustment factor is updated at least annually as described in the recommendation presented in Section 5.4 below.

Irrespective of these past flow-measurement improvements and I&I reductions, the marked decrease in plant flow over the last four years was largely due to another factor, water conservation. At less than 1 MGD, the reported flows were the lowest on record, with 2016 being the lowest by far at 0.84 MGD. Some of this recent flow decrease can be attributed to reduced groundwater infiltration into the collection

system due to a lowering of the water table during the prolonged drought. However, much of the decrease was undoubtedly related to the successful water-conservation measures implemented by the citizens of Cayucos and Morro Bay. For example, the City of Morro Bay reduced water usage during 2015 by 13.5%, significantly surpassing the 12% mandatory water-restriction goal imposed by statewide limits that went into effect in April of that year.

5.1.3 Effluent Constituents

The treatment process was designed to remove organic particulates from the wastewater stream and disinfect effluent. As with most municipal treatment plants, the MBCSD WWTP was not designed to eliminate chemical contaminants that may be present within the wastewater entering the plant. Instead, a vigorous pollution-prevention program is in place, which aims to limit the introduction of chemical contaminants at the source, before they enter the collection system. The multifaceted pollution-prevention program includes public education efforts, an onsite hazardous waste collection facility, source identification, a pharmaceutical take-back program, and inspections of commercial and industrial users. Domestic users generate more than 80% of the sewage processed at the WWTP; non-industrial users or light industry, which generate wastewater similar to that of domestic sources but on a larger scale, contribute the remaining portion of the WWTP's influent. In the absence of heavy industry within the service area, there is a concomitant lack of industrial pollutants within the MBCSD wastewater.

As in prior years, a few common metals and ubiquitous chemical compounds appeared in low concentrations within the effluent and biosolid samples collected during 2016. Of the 78 chemical compounds tested for in the semi-annual effluent samples, only a few were present in quantifiable amounts. Additionally, the measured concentrations of these compounds were all well below applicable NPDES discharge limits. In most cases, the concentrations were orders of magnitude lower than their respective limits. The associated mass emissions were also well below the limits identified in the discharge permit.

Quantifiable compounds within the effluent included three commonly occurring trace metals (copper, lead, and zinc), arsenic, selenium, and radionuclides. The three trace metals all occur naturally within the mineralogy of sediments along the central California coast. These metals also enter the collection system through internal corrosion of household plumbing systems. The two metalloids, arsenic and selenium, also occur naturally within the natural mineralogy of the region, but are less likely to arise within plumbing systems. Lastly, some level of radioactivity in effluent samples is expected because of naturally occurring radionuclides. Nonetheless, radiation levels were still well within the limits established for drinking water standards and were similar to historical levels.

Bioassays conducted over the past two decades have consistently demonstrated the MBCSD effluent's low toxicity to marine organisms. The chronic bioassays conducted during 2016 again confirmed the benign nature of MBCSD effluent. Average toxicity levels during 2016 were seven-times lower than the permitted limit.

As with effluent samples, chemical analyses of biosolid samples quantified only very low concentrations of some commonly occurring wastewater constituents. In addition to the bulk organic compounds, the 15 constituents with quantifiable concentrations consisted of ubiquitous metals, metalloids, and cyanide. All measured concentrations were well below regulatory limits that would make the biosolids hazardous or unsuitable for composting and land application.

5.2 RECEIVING WATERS

The receiving-water environment was monitored on a quarterly basis to evaluate the oceanographic conditions near the outfall, particularly with respect to any adverse impacts from wastewater discharge. Comparisons of water quality at the boundary of the zone of initial dilution (ZID) with gradient areas beyond the dilution zone documented compliance with the receiving-water objectives of the California Ocean Plan (COP) as specified in the NPDES discharge permit. Extremely sensitive electronic probes provided a detailed picture of water quality during the four surveys conducted during 2016. Precision navigation, in combination with high-resolution data on light transmittance, density, temperature, salinity, pH, and dissolved oxygen delineated the limited spatial extent of the dilute effluent plume within receiving waters.

During all four surveys, small anomalies in water properties associated with the submerged wastewater plume were detected. In all cases, the water-quality fluctuations were restricted to the ZID, were generated by the upward displacement of ambient seawater and not the presence of wastewater constituents, or were insignificant compared to the larger ambient variations resulting from natural oceanographic processes. Several of these plume measurements captured the signature of wastewater while it was undergoing rapid initial mixing within the ejection jet emanating from a diffuser port. Dilution rates determined from these close-in measurements were compared with expected critical initial dilution ratios based on modeling used to design the outfall. They demonstrated that the diffuser structure had dispersed the wastewater to a much greater extent than predicted by the modeling.

Because tests for compliance with the receiving-water limitations in the discharge permit only apply outside the ZID, these close-in measurements were not subject to the standards in the COP. Nevertheless, plume observations collected within the ZID were routinely below the permit limits applicable to observations collected outside this narrow 15-m mixing zone. None of the observed conditions suggested that unmixed wastewater was tangibly affecting receiving waters within or beyond the ZID. As with prior monitoring, water quality parameters measured during the 2016 surveys confirmed that the diffuser was operating efficiently and that wastewater was diluted 220-fold immediately following discharge from the diffuser ports and well before the completion of the initial-dilution process.

5.3 SEAFLOOR SEDIMENTS

The monitoring program has evaluated physical, chemical, and biological conditions within the benthic sediments around the outfall for more than three decades. Temporal fluctuations in these factors have been directly related to natural influences, the largest of which occur on seasonal and interannual time scales. The more notable faunal variations have typically involved interannual population fluxes within individual taxa. For example, substantial increases in the abundance of juvenile Pacific sand dollars (*Dendraster excentricus*) were documented in 1989, 1991, 1999, and 2009. During and after these episodic recruitment events, sand dollar populations overwhelmingly dominated the infaunal community, with residual effects from the initial successful recruitment persisting over a few of the following years. However, the residual effect of the 2009 sand dollar recruitment was more prolonged, and extended into 2016. The marked population increases in sand dollars have often coincided with the well-recognized global climate fluctuation known as El Niño. These major changes affect all of the benthic monitoring stations, including the distant reference site, and are related to inherent oceanographic variability rather than the discharge of effluent.

To test for outfall-related effects, a large number of biological indices and parameters were computed from an enumeration of the 262,000 specimens collected over 31 years of benthic monitoring. None of

these parameters exhibited statistically significant spatial distributions related to the effluent discharge or long-term spatiotemporal trends indicative of an increasingly degraded benthic habitat near the outfall.

Not only were spatial differences found in individual surveys small compared to inherent sampling variability, but also the differences were generally smaller than seasonal and interannual changes in community structure that arose from natural environmental oscillations, such as El Niño. Despite large temporal fluctuations in the composition of the infaunal community, its health has remained consistently high in the 29 years of monitoring prior to the 2015 survey. During that time, the benthic environment within the survey area around the outfall was populated by an infaunal community dominated by pollution-sensitive suspension-feeding organisms that thrive in sediments nearly devoid of organic material.

However, a major change occurred during 2015 when numerous large sand dollars totally dominated every aspect of the sedimentary environment throughout the offshore survey area. This population of sand dollars originated with the vast number of juveniles that were recruited during the 2009 El Niño event. As this original 2009-cohort matured over the following five years, sand dollars displaced a steadily increasing amount of ambient surficial sediment and the infauna that resided in it. This trend culminated during 2015 with a large and abrupt decline in nearly every measure of the health of the benthic infaunal community, including infaunal density, diversity, species counts, and richness. The declines were unrelated to the MBCSD discharge because they occurred uniformly throughout the survey area with no evidence of a spatial gradient related to outfall proximity.

The most dramatic decline occurred in the infaunal trophic index (*ITI*), which estimates the wellbeing of the infaunal community from the relative sizes of the suspension-feeding infaunal population, which typically reside in clean sediments, and the pollution-tolerant detritus-feeding population, which opportunistically occupy organically enriched seafloor habitats. Prior to improvements in wastewater treatment during the 1980s, marked *ITI* declines were observed within seafloor sediments immediately surrounding large ocean discharges in other regions. However, the *ITI* decline measured in the MBCSD survey area during 2015, and that continued into 2016, was unrelated to organic loading within sediments because the concentrations of organic constituents measured during 2015 and 2016 were comparable to those of prior years.

Instead, the *ITI* decline was directly due to the development of the mature sand dollar bed that eliminated nearly all suspension-feeding infauna through predation, and competition for space and food. In addition, the large sand dollar bed provided a suitable habitat for a number of parasitic and opportunistic species not typically seen in the MBCSD samples. In fact, two species of crab were found during the 2015 and 2016 surveys that were previously unseen in the MBCSD database, which is remarkable considering 262,000 specimens representing 392 individual taxa have been collected over the 31 years of monitoring. Some of the shift in the infaunal population that weighted heavily in the observed *ITI* reduction during 2015 and 2016 was unrelated to habitat degradation from an accumulation of organic contaminants. Field studies conducted on uncontaminated sediments in other regions confirmed that many of those same species were far more prevalent within sand dollar beds than in seafloor areas immediately adjacent to the beds. Clearly, the presence of the mature sand dollar bed had a profound effect on the benthic environment within the MBCSD survey area beginning in 2015, and extending into 2016.

Beginning with the 2016 survey, however, another identifiable change in the infaunal community became apparent. An assemblage of detritus-feeding organisms that thrive in organically enriched sediments became prevalent in several isolated grab samples. The samples were not collected in proximity to the outfall, so the wastewater discharge was not the source of the organic material. Instead, these grab samples also contained dead sand dollars in numbers that had not been seen in prior surveys, or that were

present in most of the other samples collected during the 2016 survey. After six years, the original 2009 cohort of sand dollars had begun to die off. As the senescence of the original 2009 sand-dollar cohort continues into future years, the increased amount of decaying organic material they contribute to the sediments of northern Estero Bay will trigger a more widespread shift in the resident infaunal population to one dominated by opportunistic detritus feeders.

Spatiotemporal analyses were also conducted on the large dataset of sediment physicochemistry spanning the 31 years since benthic monitoring began in 1986. Those analyses demonstrated that there was no buildup of sediment contaminants surrounding the outfall. In addition, trace-metal and organic concentrations collected both near and far from the outfall during 2016 remained below thresholds considered harmful to marine biota. In fact, sediment trace-metal concentrations within Estero Bay were well below concentrations found in the vast majority of samples collected offshore Southern California. This attests to the pristine nature of the ocean environment in northern Estero Bay. Nickel and chromium are the only trace metals with comparatively elevated concentrations within Estero Bay sediments. However, the concentrations of these particular metals are naturally elevated in the chromite mineral ores found in the region. As a result, the Estero Bay sediment metal concentrations are comparable to those in nearby benthic environments, such as Port San Luis (NOAA 1991a), and the Morro Bay Estuary (Tenera and Marine Research Specialists 1997).

These sediment-chemistry analyses, in conjunction with analyses of infaunal community structure, provide strong empirical evidence that the sediments surrounding the outfall, along with the organisms within them, have not been perceptibly impacted by the discharge, even after more than three decades of plant operation and discharge monitoring.

5.4 RECOMMENDATIONS

The 31 years of monitoring data summarized in this annual report demonstrate that the MBCSD WWTP has been operating as designed, and that the discharge of effluent has not adversely affected the marine environment within Estero Bay. Occasional adjustments made to the monitoring program have substantially increased its capacity to detect minute environmental impacts, yet conclusions regarding the lack of impacts have continued to hold true into 2016.

Past adjustments to the monitoring program were largely based on recommendations for improving its ability to detect impacts and assess compliance. Few recommendations have dealt with changes to the treatment plant or its process, largely because of the consistently high performance of the plant and outfall system. Thus, as in past reports, most of the following recommendations pertain to proposed revisions to the monitoring program:

- **Reduce the monitoring frequency for effluent chemical parameters.** Pursuant to the provisions within the current NPDES permit, a reduction in the monitoring frequency of effluent chemical concentrations is warranted (MBCSD 2004; MRS 2004b, 2009b). The MBCSD discharge permit allows a change in monitoring frequency to once-in-the-life of the permit for those compounds that are undetected in the first year of monitoring. During 2009 sampling, which was the first year of sampling under the current permit, 73 chemical constituents met this criterion. Additionally, application of the SWRCB's (2005) statistical methodology to two decades of MBCSD effluent measurements unequivocally demonstrates that there is no reasonable potential for exceedance of permit limits for those 73 compounds, or for the remaining compounds currently monitored on an annual basis. As a result, the COP does not require continued annual monitoring of any of the compounds. Concentrations measured in effluent samples collected in subsequent years, including 2016, only serve to strengthen those conclusions. The reduction in monitoring frequency is further

justified by the historically high level of compliance, a proposed monitoring frequency that is consistent with other similar-sized dischargers, and an ongoing bioassay program that provides a more sensitive all-inclusive evaluation of effluent quality.

- **Reinstate analysis for trace metals, grain size, and infauna at benthic Station B1.** Although the physical environment at Station B1 departs from that at the outfall stations, its 1 km distance from the outfall provides valuable insight into large-scale environmental processes within Estero Bay that are independent of the discharge. Additionally, Station B1 has been continuously sampled throughout the 31-year monitoring program. Even though benthic monitoring at this location is no longer required by the current discharge permit, voluntary collection and analysis of sediment samples has continued through 2016. Continued sampling at this location is recommended to maintain historical consistency with the long-term time series, and to provide insight into regional processes affecting Estero Bay as a whole.
- **Eliminate the Cat-Litter Public-Outreach Program.** The current discharge permit requires an annual reevaluation of the implementation goals of the Cat-Litter Public-Outreach work plan. A requirement for development and submittal of a work plan was included in the current permit as a conservation measure because concerns were raised that Estero Bay was a “hot-spot” for otter mortality associated with *T. gondii* infection. Disposal of cat litter into the MBCSD collection system was thought by some to be a contributing factor to the high exposure and infection rates found in live animals and carcasses tested from the Estero Bay area. However, shortly after final approval of the current MBCSD permit in 2009, results from a comprehensive field study (Johnson et al. 2009) were published that confirmed that disease vectors unrelated to WWTP discharge are responsible for the observed *T. gondii* exposure in otters, and that the epicenter for infection is not within Estero Bay. As such, there is little scientific rationale for the continuation of a dedicated outreach program specific to cat-litter disposal in the MBCSD collection system.

Instead, existing public-outreach efforts required under the permit should continue to incorporate efforts to keep regulatory agencies and the public abreast of the latest scientific findings in order to reduce existing misconceptions about the MBCSD discharge’s effects on otter health and the significance of *T. gondii* within the watershed. Nevertheless, although the current MBCSD efforts at public education and outreach regarding cat litter disposal are comprehensive, and have included the use of the internet, posters, newsletters, and public talks and tours, they remain uncodedified by a formal work plan, as required in the current NPDES permit. Therefore, until the RWQCB revisits the validity of the concerns underlying the inclusion and necessity of this conservation measure, the MBCSD should continue to work towards finalizing a cat-litter public-outreach work plan in accordance with the permit requirements.

- **Remove Basin-Plan receiving-water limits on pH and DO from the NPDES discharge permit.** As discussed in Section 3.3.2, the Basin Plan limits on pH and DO that are incorporated into the current NPDES discharge permit issued to the MBCSD are inappropriate for open ocean dischargers. The fixed Basin Plan limits were largely designed for discharges to onshore surface waters, where there is little natural variation in pH and DO within the receiving waters. Conversely, natural oceanographic processes, such as upwelling, regularly cause the DO and pH within the ambient receiving water surrounding the MBCSD outfall to range beyond the Basin Plan limits. In contrast to the Basin Plan limits, the COP recognizes the potential for inherent variation in the receiving-water characteristics and specifies limits on excursions in these two water properties relative to background levels present at the time of the survey. Because the COP receiving-water objectives are designed to be adequately protective of the marine environment, application of the fixed Basin Plan limits to the

same receiving-water characteristics already covered by the COP is not only redundant but also inappropriate.

- **Remove the effluent nutrient-monitoring requirement.** A provision for nutrient monitoring was incorporated into the current discharge permit to address concerns regarding the MBCSD's potential nutrient contribution to the generation of harmful algal blooms offshore central California. As discussed in Section 2.2.11, however, chemical analyses conducted to date demonstrate that nutrient concentrations within the MBCSD effluent, and mass loading to the marine environment from its discharge, are insignificant compared to both other central-coast dischargers, and the contribution from regional streams and rivers. Specifically, nutrient loading from the MBCSD WWTP is several orders-of-magnitude lower than both runoff and discharge from other central-coast WWTPs, and far smaller than the nutrient loading from naturally occurring processes such as upwelling. Additionally, it is clear that nutrient loads from the MBCSD discharge are unrelated to the frequency or intensity of the algal bloom occurring along this stretch of coastline. Consequently, continued nutrient monitoring provides no scientifically valid or usable information relevant to the prediction or management of algal blooms, and should be discontinued.
- **Each year, establish an annual schedule for BOD sampling that addresses all three of the competing permit requirements governing collection frequency. Clarify BOD sampling requirements and correct associated footnote errors in future discharge permits issued to the MBCSD.** Under the current permit, there are three separate requirements dealing with the collection of weekly effluent samples for BOD analysis:
 1. The first requirement for 'weekly' sampling stipulates that one sample must be collected during each complete week of the year. Presumably, each calendar week extends from Sunday through Saturday as indicated by the definition of the *Average Weekly Effluent Limitation* on *Page A-1* of the permit. This requirement results in the collection of approximately 52 BOD samples each year, except when the calendar week extends into a prior or future year, and the weekly sample is collected on a day that falls outside of the current year. In such cases, a given year may have as many as 54 'weekly' samples, or as few as 50 samples.
 2. The second sampling requirement arises from the definition of a monthly average in the *Central Coast Standard Provisions* that are incorporated in the NPDES permit (*Page D-20*). It requires the use of at least four sample-results to compute a monthly average.
 3. Interpretation of the third weekly sampling requirement is confounded by the incorrect application of *Footnote 1* to the sampling frequency requirement for effluent BOD that is listed on *Page E-6* of the NPDES Permit. That footnote refers to influent, rather than effluent sampling. Nevertheless, assuming that the correct footnote for weekly BOD sampling is *Footnote 2*, each day of the 7-day week must be represented in each two-month period. This is typically achieved by rotating the day of the week used for sampling and by duplicating sampling on one day of the week once during each eight-week period.

Designing a sampling schedule, which accomplishes all three of these requirements efficiently, is a complex process. For example, application of only the first or third sampling requirements may occasionally result in less than four samples being drawn in a given month. Advance preparation of an annual schedule for BOD sampling during 2017 will help to achieve the permit requirements efficiently while minimizing extraneous sampling.

- **Explicitly state that the 75% removal requirement only applies when influent TSS concentrations exceed 240 mg/L.** The NPDES discharge permit states that “*The Discharger shall, as a 30-day average, remove at least 75% of suspended solids and 30% of BOD₅ from the influent stream before discharging wastewater to the ocean, except that the limit shall not be less than 60 mg/L*” (RWQCB-USEPA 2009).¹ The statement is unnecessarily confusing and makes quantitative evaluation of compliance difficult. Specifically, the statement does not make it clear that the applicability of the removal-rate requirements is conditional, and therefore does not apply in certain monthly compliance evaluations.

The limitation derives from the first note to Table A of the COP (SWRCB 2005), which makes it clear that the applicability of the requirement is conditional: “...*the discharger shall remove 75% of suspended solids from the influent stream at any time the influent concentration exceeds four times [the lower effluent concentration limit].*” In the case of the MBCSD discharge, the effluent TSS threshold of applicability is 60 mg/L, so the influent TSS threshold is 240 mg/L. Because the target BOD removal-rate is 30%, the influent BOD threshold of applicability is 86 mg/L. Although the MBCSD TSS and BOD influent concentrations rarely drop below these thresholds of applicability, the discharge permit should, at a minimum, explicitly state that the removal-rate requirements are conditional on exceedance of the influent thresholds identified above. For TSS, the permit requirement should read: *When monthly average influent TSS concentrations exceed 240 mg/L, the discharger shall remove at least 75% of the solids from the influent stream.*

In practice, MBCSD effluent TSS and BOD concentrations are almost always below 60 mg/L, and when this is the case, compliance with the conditional removal-rate requirement is guaranteed. Specifically, if the sub-60-mg/L effluent concentration was not achieved by removal exceeding the 75%/30% requirements, then the requirements do not apply, because the influent concentration must have been less than the 240 mg/L or 86 mg/L thresholds to begin with.

- **Apply a correction formula when reporting daily flow using effluent flow-meter measurements.** Daily plant throughput is normally determined by totalizing the readings from the influent flow meter. The influent flow-meter readings are used because they measure the volume of wastewater processed by the treatment plant more accurately than the effluent flow meter. On rare occasions, however, the influent flow meter reports erroneously high values. Typically, these outliers are obvious in the flow record and result from easily identifiable causes such as surcharging of the influent trunk line during major rain events, when equipment malfunctions occur at the headworks, or, as in 2016, when inflow to the WWTP is intentionally stopped and allowed to backup in the trunk line to facilitate equipment repair. On those occasions, daily flow should be reported using measurements from the effluent flow meter that have been adjusted downward to account for that meter’s overtotalization.

Historically, the effluent flow meter has been found to overtotalize the actual flow by approximately 25% [MRS 2003b]. A comparison of daily flow totals reported by the influent and effluent meters during 2016 (Figure 5.1), however, provides an updated formula for adjusting the effluent flow-meter readings that is more representative of current conditions:

$$A = 0.78 \times E - 0.08 \quad \text{Equation 5.1}$$

where: A = the actual plant flow in MGD, and
 E = the flow measured by the effluent meter.

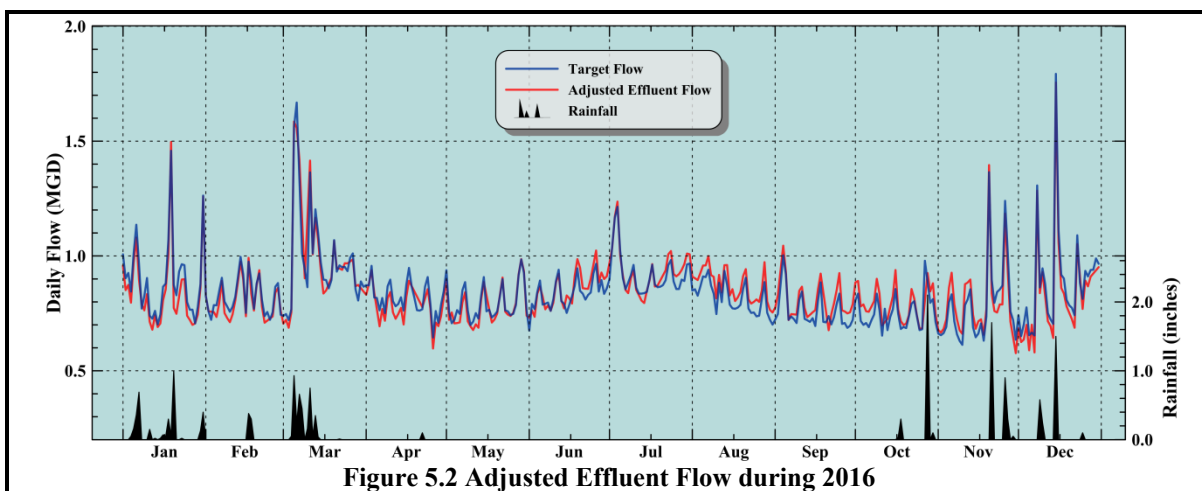
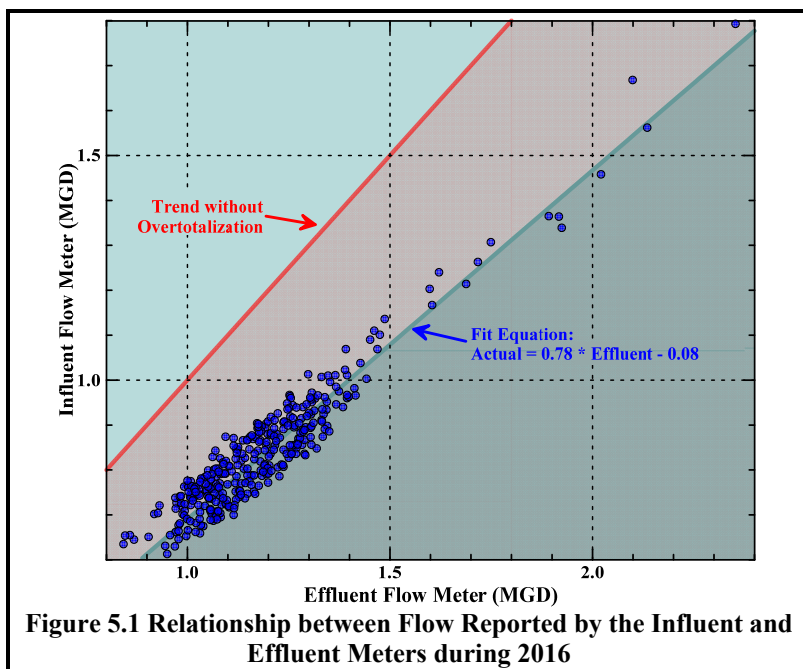
¹ Section IV.B on Page 16 of the NPDES discharge permit

Equation 5.1 was determined from a linear regression on 338 pairs of daily flow observations that were measured by the influent and effluent meters throughout 2016 (blue dots in Figure 5.1). Missing or anomalous influent measurements associated with surcharging in the trunk line were excluded in the regression analysis.

The relationship between influent and effluent readings changes slowly over time due to inherent drift in the meters, or sporadically when either meter is recalibrated, serviced, or replaced. Nevertheless, while updating the adjustment

equation annually is useful, the actual interannual changes are small compared to the overall adjustment for overttotalization of the effluent meter. Specifically, the red line in Figure 5.1 displays the relationship between the influent and effluent meters that would be expected in the absence of overttotalization by the effluent meter, namely, exactly the same readings from each meter. That relationship departs significantly from the scatter among the cloud of dark-blue points surrounding the green regression line in the Figure, which represent actual measurements made by each meter during 2016.

Occasionally replacing an obviously errant influent meter reading with an adjusted effluent-meter reading is a justifiable approach for establishing a continuous time series of reliable flow data. Daily flow predicted by adjusting the effluent flow-meter data using Equation 5.1 (red time series in Figure 5.2) shows excellent agreement with influent data (blue time series) throughout 2016, and lends confidence in its use when the influent meter becomes surcharged. Normally, the influent



measurements, which are based on precision water-level measurements, provide a more reliable determination of flow than the impeller-based effluent meter. However, when readings from the influent meter are obviously compromised, replacing those erroneous daily flow values with an adjusted effluent reading provides a credible estimate of the flow.

The availability of precise and closely matching measurements from both the influent meter and the adjusted effluent meter provides valuable redundancy in the event that one of the meters is taken offline for repairs or calibration. However, to guarantee reporting of the most accurate daily flow totals on a regular basis, future NPDES discharge permits issued to the MBCSD should explicitly state that flow data from either the influent meter or effluent meter (after adjustment) may be used to establish the daily flow volumes. To ensure that adjustments to effluent flow readings continue to be as accurate and timely as possible, the coefficients in Equation 5.1 should be recomputed at least annually, or when major modifications are made to the meters or flow system.

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Table A.1 WWTP Specifications

Parameter	Quantity	Parameter	Quantity
Waste Loading		Tank 2	
Flow (MGD)		Surface loading rate PSDWF	2.74
Average dry-weather flow	2.06	(10 ³ L/m ² /day)	29.74
PSDWF	2.36	Detention time at PWWF (hr.)	2.2
Peak dry-weather flow	6.64	Total Treatment	
PWWF	6.60	Overall treatment efficiencies (%)	
Strength		BOD ₅	57
BOD ₅ (mg/L)	280	Suspended solids	75
Suspended solids (mg/L)	280	Expected effluent quality (mg/L)	
Grit (ft ³ /mg)	10	BOD ₅	120
Waste quantities at PSDWF		Suspended solids	70
BOD ₅ (mt/day)	2.5	Solids stabilization	
Suspended Solids (mt/day)	2.5	Anaerobic digester loading (mt/day)	
Grit (ft ³ /day)	23.6	Primary solids	1.6
Preliminary Treatment		Secondary solids	0.4
Mechanically Cleaned Bar Screen		Assumed sludge volatile content (%)	
Number	1	Primary solids	70
Capacity (MGD)	8.2	Secondary solids	82
Channel Monster		Sludge volume (m ³ /day)	50.7
Number	1	Digester 1 (existing, fixed cover)	
Capacity (MGD)	7.0	Diameter (m)	12.2
Influent Pumps (variable speed)		Side water depth (m)	4.9
Number	3	Volume (m ³)	629
Capacity each (MGD)	3.3	Digester 2 (existing, fixed cover)	
Total head (m)	9.6	Diameter (m)	12.2
Aerated Grit-Removal Tanks		Side water depth (m)	5.8
Number	1	Volume (m ³)	725
Length (m)	9.1	Digester 3 (new, floating cover)	
Width (m)	4.9	Diameter (m)	10.7
Depth (m)	2.4	Side water depth (m)	6.9
Detention time at PWWF (min)	6.3	Volume (m ³)	646
Grit Pumps		Hydraulic detention time based on	
Number	2	net volume of digesters 2 and 3	23
Capacity (gpm)	250	(days)	
Primary Treatment		Assumed volatile solids reduction	
Sedimentation Tanks		(%)	55
Number	2	Expected sludge gas production	804
Diameter (m)		(m ³ /day)	
Tank 1	15.2	Sludge Drying Beds	
Tank 2	12.2	Number	12
Average side water depth (m)		Length each (m)	49.4
Tank 1	2.74	Width each (m)	9.8
		Solids Loadings (kg ft ⁻¹ yr ⁻¹)	78.3

Parameter	Quantity
Assumed removal efficiency (%)	
BOD ₅	35
Suspended solids	65
Primary effluent quality (mg/L)	
BOD ₅	182
Suspended solids	98
Secondary treatment	
Biofilters (existing, in partial secondary treatment mode of operation)	
Flow distribution at PSDWF (MGD)	
Biofilter 1	0.39
Biofilter 2	0.58
Diameter (m)	
Biofilter 1	18.3
Biofilter 2	21.3
Net media surface area (m ²)	
Biofilter 1	262
Biofilter 2	350
Average media height (m)	
Biofilter 1	1.4
Biofilter 2	1.5
Media Volume (m ³)	
Biofilter 1	360
Biofilter 2	532
Specific organic loading rate (lbs BOD ₅ /day/1000 ft ³)	47
Circulated flow (MGD)	
Biofilter 1	1.37
Biofilter 2	2.04
Hydraulic loading rate (gpm/ft ² media surface)	
Biofilter 1	0.34
Biofilter 2	0.38
Circulation Pumps	
Biofilter 1	
Capacity (gpm)	950
Total head (m)	3.4
Biofilter 2	
Capacity (gpm)	1420
Total head (m)	4.3
Stand-by (2-speed)	
Capacity (gpm)	960
	1660
Total head (m)	3.4
	4.4

Parameter	Quantity
Interstage pumping	
Biofilter Effluent Pumps (variable speed)	
Number	2
Capacity each (gpm)	2300
Total head (m)	8.2
Secondary sedimentation Tanks	
Number	1
Diameter (m)	16.8
Tank surface area (m ²)	221
Tank volume (m ³)	3125
Average water depth (m)	4.6
Overflow rate at PSDWF (10 ³ L/m ² /day)	16.6
Expected secondary treatment effluent quality (mg/L)	
BOD ₅	30
Suspended Solids	30
Chlorination	
Chlorine Contact Tank (existing)	
Number of passes	2
Length (m)	
Pass 1	16.8
Pass 2	22.9
Width each pass (m)	4.6
Average depth (m)	2.3
Total volume (m ³)	413
Detention time at PDWF (min)	24
Chlorinators	
Pre-chlorinator	
Number	1
Initial capacity (kg/day)	227.3
Ultimate capacity (kg/day)	909.1
RAS chlorinator	
Number	1
Capacity (kg/day)	227.3
Ultimate capacity (kg/day)	90.9
Sodium Hypochlorite Post Chlorinator	
Chemical feed pumps	3
Combined Capacity (kg/day)	5450
Dechlorination	
Sodium Bisulfite System	
Chemical feed pumps	3
Combined Capacity (kg/day)	1226

INFAUNAL INDICES

B.1.1 Species Diversity

The Shannon diversity index (H') (Shannon and Weaver 1949) is the most common diversity index used in 301(h) programs. It measures the relative distribution of individual organisms among the species present in the sample. H' increases for broader distributions of individuals among species. If all individuals are of one species then H' is 0.00. If each individual organism is a separate species then H' is determined by the logarithm of the total number of organisms collected. For other distributions, the

diversity index is given by: $H' = - \sum_{j=1}^S \left[\left(\frac{n_j}{N} \right) \ln \left(\frac{n_j}{N} \right) \right]$; where: S = total number of species, n_j = number of individuals in the j^{th} species, N = total number of individuals, and $\ln$ = natural logarithm (base e). The value of the Shannon index usually falls between 1.5 and 3.5 and only rarely surpasses 4.5 (Margalef 1972).

Some studies have found a decrease in this index in response to pollutant stress in benthic communities. However, this index is ambiguous because it depends on how evenly the organisms are distributed among the species. Consequently, a statistically significant reduction in this index can occur in the absence of anthropogenic (human-induced) stresses.

When randomness in the sample cannot be guaranteed, then the Brillouin index (h) is a more appropriate

measure of diversity (Pielou 1977). It is calculated using the formula: $h = \frac{\ln(N!) - \sum_{j=1}^S [\ln(n_j!)]}{N}$. It gives a similar measure of diversity although it produces a slightly lower value than the Shannon index when applied to the same data. In contrast to the Shannon index, the Brillouin will vary between samples although the number of species and their proportional abundance remain the same. Because of its dependence on sample size and the increased complexity of its computation, the Brillouin index is more rarely used than the Shannon index.

B.1.2 Evenness

The Pielou evenness index (J') (Pielou 1977) measures how evenly the individual organisms are distributed among the species present in the sample. J' increases for more even distributions of individuals among species. If all individuals belong to a single species, then J' is indeterminate. If each species is represented by a single organism, then J' will be equal to 1.00. For other distributions, the evenness index is given by: $J' = \frac{H'}{\ln S}$; where: S = total number of species, and H' = Shannon-Wiener diversity index.

Because this index is derived from the Shannon-Wiener diversity index (H'), it is subject to the same limitations.

B.1.3 Species Dominance

Dominance indices increase with decreasing diversity and are heavily weighted toward the most common species in a sample. The best known of these measures is Simpson's index (C') (Simpson 1949; Wittaker 1965). It is related to other diversity and evenness indices but increases with increasing proportions of individuals associated with a few species. If all individuals are of one species, then C' is maximum and equal to the maximum dominance of 1.00. If individual organisms are evenly distributed among species

($J'=1.00$), then C' asymptotically approaches 0.00 with increasing numbers of individuals. The Simpson dominance measure is given by: $C' = \sum_{j=1}^S \left(\frac{n_j}{N} \right)^2$ but for finite communities, the unbiased form is:

$$C' = \sum_{j=1}^S \left(\frac{n_j(n_j-1)}{N(N-1)} \right) \text{ (Magurran 1988).}$$

An unrelated measure of dominance has been ascribed to Swartz et al. (1985). The Swartz dominance index (S_w) is defined as the minimum number of species that account for at least 75% of all individual organisms collected in a sample. In this dominance measure, species are first ranked by the number of individual organisms before summing the number of species that represent a cumulative percent equal to, or first exceeding 75%. This is an inverse measure because higher dominance is reflected in a lower number of species accounting for 75% of the individual organisms. Despite this, it is not subject to many of the limitations that plague the other univariate indices. It is a non-parametric measure that does not assume an underlying distribution of individuals among species.

B.1.4 Infaunal Trophic Index

The Infaunal Trophic Index (ITI) (Word 1978) compares the abundance of four soft-bottom benthic assemblages distinguished by feeding behavior. Because the sensitivity or tolerance to organic enrichment in particulate matter differs among the four groups, shifts in group dominance, as reflected in a changing infaunal index, can be indicative of changed or degraded environmental conditions. The ITI ranges between 0 and 100. When species in Group I (suspension feeders) and Group II (surface-detritus feeders) dominate, index values are above 58 and sediments are relatively clean. Lower infaunal indices occur when species in Group III (surface deposit feeders) and Group IV (sub-surface detritus feeders) dominate and sediments are high in organics. The ITI is computed from: $ITI = 100 - 33.3 \left[\frac{G_2 + 2G_3 + 3G_4}{G_1 + G_2 + G_3 + G_4} \right]$; where:

G_i = number of individual organisms within the i^{th} trophic group.

B.1.5 Species Richness

The Margalef species richness index (d) (Margalef 1951) measures the number of species in a sample relative to the number of individual organisms. d strongly increases for increasing number of species and increases only logarithmically for decreasing number of individuals. If only one species is present then d is 0.00. For other distributions, the richness index is given by: $d = \frac{S-1}{\ln N}$. Its applicability to biological communities is dependent on whether specimens are log-normally distributed among the species in a given sample. Such an assumption is not globally applicable to benthic marine communities, and without testing each data set, the richness index is of questionable value.

Table B.1 Station Abundance by Taxa during October 2016

Taxonomic Group	Taxon	Station							Station Total	Percent of Total	Cumulative %
		B1	B2	B3	B4	B5	B6	B7			
Echinodermata	<i>Dendraster excentricus</i>	338	266	220	185	174	161	238	1582	—	—
Mollusca	<i>Balcis rutila</i>	200	40	74	115	133	210	97	869	34.16	34.16
Annelida	<i>Magelona sacculata</i>	12	37	25	89	73	53	55	344	13.52	47.68
Crustacea	<i>Rhepoxynius menziesi</i>	5	26	28	30	61	31	30	211	8.29	55.97
Nemertea	<i>Carinoma mutabilis</i>	55	16	22	20	22	17	14	166	6.53	62.50
Annelida	<i>Apoprionospio pygmaea</i>	2	26	7	25	16	39	33	148	5.82	68.32
Annelida	<i>Armandia bioculata</i>	11	4	9	26	10	44	25	129	5.07	73.39
Bivalvia	<i>Tellina modesta</i>	42	6	10	4	6	5	7	80	3.14	76.53
Nemertea	<i>Cerebratulus californiensis</i>	10	14	12	18	10	9	7	80	3.14	79.68
Annelida	<i>Glycinde armigera</i>	6	6	6	18	11	13	13	73	2.87	82.55
Crustacea	<i>Americhilidium shoemakeri</i>	14	31	15	4	1	3	1	69	2.71	85.26
Bivalvia	<i>Tellina bodegensis</i>	4	4	7	7	11	6	7	46	1.81	87.07
Annelida	<i>Sigalion spinosa</i>	5	2	4	8	3	7	11	40	1.57	88.64
Annelida	<i>Nephtys caecoides</i>	3	6	2	3	3	5	5	27	1.06	89.70
Crustacea	<i>Rhepoxynius abronius</i>	1	5	6	5	6	1	—	24	0.94	90.64
Annelida	<i>Scoloplos armiger</i>	3	—	5	5	1	3	3	20	0.79	91.43
Bivalvia	<i>Cooperella subdiaphana</i>	7	1	—	4	2	2	3	19	0.75	92.18
Nemertea	<i>Paranemertes sp. A</i>	3	—	—	1	2	5	2	13	0.51	92.69
Crustacea	<i>Cancer gracilis</i>	2	1	1	1	2	2	4	13	0.51	93.20
Annelida	<i>Lumbrineris californiensis</i>	3	1	2	1	1	—	3	11	0.43	93.63
Annelida	<i>Glycera convoluta</i>	3	—	1	1	2	3	1	11	0.43	94.06
Annelida	<i>Phyllodoce</i>	1	2	—	1	2	4	1	11	0.43	94.50
Annelida	<i>Gyptis brevipalpa</i>	4	—	—	2	3	1	—	10	0.39	94.89
Nemertea	<i>Tubulanus</i>	3	—	—	2	1	3	1	10	0.39	95.28
Annelida	<i>Prionospio cirrifera</i>	8	—	—	—	—	1	—	9	0.35	95.64
Crustacea	<i>Foxiphalus xiximeus</i>	5	—	—	2	—	1	—	8	0.31	95.95
Crustacea	<i>Gibberosus myersi</i>	3	3	1	1	—	—	—	8	0.31	96.27
Annelida	<i>Lumbrineris</i>	3	1	1	1	1	—	1	8	0.31	96.58
Crustacea	Hippolytidae	1	2	—	2	—	1	2	8	0.31	96.89
Annelida	<i>Capitella capitata</i>	—	—	—	—	—	3	5	8	0.31	97.21
Crustacea	<i>Cancer productus</i>	1	—	1	—	—	5	—	7	0.28	97.48
Annelida	<i>Chaetozona setosa</i>	1	—	—	1	3	—	2	7	0.28	97.76
Bivalvia	<i>Siliqua lucida</i>	2	—	—	—	1	2	1	6	0.24	98.00
Bivalvia	<i>Rochefortia tumida</i>	—	2	—	1	1	—	1	5	0.20	98.19
Annelida	<i>Diopatra ornata</i>	—	1	—	1	1	1	—	4	0.16	98.35
Annelida	<i>Syllides</i>	—	—	1	2	—	—	1	4	0.16	98.51
Annelida	<i>Eumida</i>	—	1	—	—	—	2	—	3	0.12	98.62
Mollusca	<i>Alvania</i>	—	1	—	—	—	1	1	3	0.12	98.74
Nemertea	<i>Lineus bilineatus</i>	—	—	—	1	1	—	1	3	0.12	98.86
Echinodermata	<i>Leptosynapta</i>	—	—	—	—	1	2	—	3	0.12	98.98
Platyhelminthes	Polycladida	1	—	1	—	—	—	—	2	0.08	99.06
Bivalvia	<i>Chione</i>	—	—	2	—	—	—	—	2	0.08	99.14
Annelida	<i>Tenonia priops</i>	—	—	1	1	—	—	—	2	0.08	99.21
Crustacea	<i>Lepidopa californica</i>	—	—	—	—	1	1	—	2	0.08	99.29
Annelida	<i>Harmothoe</i>	—	—	—	—	1	1	—	2	0.08	99.37

Table B.1 Station Abundance by Taxa during October 2016

Taxonomic Group	Taxon	Station							Station Total	Percent of Total	Cumulative %
		B1	B2	B3	B4	B5	B6	B7			
Mollusca	<i>Epitonium sawinae</i>	1	–	–	–	–	–	–	1	0.04	99.41
Bivalvia	<i>Macoma acolasta</i>	1	–	–	–	–	–	–	1	0.04	99.45
Annelida	<i>Scalibregma californicum</i>	1	–	–	–	–	–	–	1	0.04	99.49
Nematoda	Nematoda	1	–	–	–	–	–	–	1	0.04	99.53
Crustacea	Cumacea	1	–	–	–	–	–	–	1	0.04	99.57
Crustacea	<i>Blepharipoda occidentalis</i>	–	1	–	–	–	–	–	1	0.04	99.61
Vertebrata	<i>Chilara taylori</i>	–	–	–	1	–	–	–	1	0.04	99.65
Vertebrata	<i>Porichthys notatus</i>	–	–	–	–	1	–	–	1	0.04	99.69
Crustacea	Aoridae	–	–	–	–	–	1	–	1	0.04	99.72
Cnidaria	Actiniaria	–	–	–	–	–	1	–	1	0.04	99.76
Annelida	<i>Pectinaria californiensis</i>	–	–	–	–	–	1	–	1	0.04	99.80
Peanut Worms	Sipuncula	–	–	–	–	–	–	1	1	0.04	99.84
Echinodermata	<i>Amphiodia</i>	–	–	–	–	–	–	1	1	0.04	99.88
Annelida	<i>Nephtys cornuta franciscana</i>	–	–	–	–	–	–	1	1	0.04	99.92
Annelida	<i>Spiophanes bombyx</i>	–	–	–	–	–	–	1	1	0.04	99.96
Annelida	<i>Ampharete labrops</i>	–	–	–	–	–	–	1	1	0.04	100.00
Total Number of Individuals		767	506	464	589	568	651	581	4126		
Total Number of Individuals Excluding <i>D. Excentricus</i>		429	240	244	404	394	490	343	2544		

Table B.2 Replicate Sample Abundance by Taxa at Station B1 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	83	100	95	30	30	338	–	–
Mollusca	<i>Balcis rutila</i>	52	61	35	23	29	200	46.62	46.62
Nemertea	<i>Carinoma mutabilis</i>	12	5	13	8	17	55	12.82	59.44
Bivalvia	<i>Tellina modesta</i>	12	10	1	9	10	42	9.79	69.23
Crustacea	<i>Americhilidium shoemakeri</i>	4	1	2	3	4	14	3.26	72.49
Annelida	<i>Magelona sacculata</i>	–	1	2	4	5	12	2.80	75.29
Annelida	<i>Armandia bioculata</i>	4	4	–	–	3	11	2.56	77.86
Nemertea	<i>Cerebratulus californiensis</i>	1	4	2	2	1	10	2.33	80.19
Annelida	<i>Prionospio cirrifera</i>	1	2	2	–	3	8	1.86	82.05
Bivalvia	<i>Cooperella subdiaphana</i>	4	2	–	–	1	7	1.63	83.68
Annelida	<i>Glycinde armigera</i>	–	–	2	2	2	6	1.40	85.08
Annelida	<i>Sigalion spinosa</i>	1	–	1	3	–	5	1.17	86.25
Crustacea	<i>Rhepoxynius menziesi</i>	2	1	1	–	1	5	1.17	87.41
Crustacea	<i>Foxiphalus xiximeus</i>	2	–	3	–	–	5	1.17	88.58
Annelida	<i>Gyptis brevipalpa</i>	–	2	1	–	1	4	0.93	89.51
Bivalvia	<i>Tellina bodegensis</i>	–	–	–	1	3	4	0.93	90.44
Annelida	<i>Glycera convoluta</i>	1	–	2	–	–	3	0.70	91.14
Annelida	<i>Lumbrineris californiensis</i>	1	–	2	–	–	3	0.70	91.84
Annelida	<i>Lumbrineris</i>	–	1	–	1	1	3	0.70	92.54
Nemertea	<i>Paranemertes sp. A</i>	–	1	–	2	–	3	0.70	93.24

Table B.2 Replicate Sample Abundance by Taxa at Station B1 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Nemertea	<i>Tubulanus</i>	–	2	–	1	–	3	0.70	93.94
Annelida	<i>Scoloplos armiger</i>	–	2	–	–	1	3	0.70	94.64
Crustacea	<i>Gibberosus myersi</i>	–	–	1	2	–	3	0.70	95.34
Annelida	<i>Nephtys caecoides</i>	–	–	–	2	1	3	0.70	96.04
Crustacea	<i>Cancer gracilis</i>	2	–	–	–	–	2	0.47	96.50
Annelida	<i>Apoprionospio pygmaea</i>	–	–	–	1	1	2	0.47	96.97
Bivalvia	<i>Siliqua lucida</i>	–	–	–	–	2	2	0.47	97.44
Annelida	<i>Phyllodoce</i>	1	–	–	–	–	1	0.23	97.67
Annelida	<i>Scalibregma californicum</i>	1	–	–	–	–	1	0.23	97.90
Crustacea	Hippolytidae	–	1	–	–	–	1	0.23	98.14
Mollusca	<i>Epitonium sawinae</i>	–	1	–	–	–	1	0.23	98.37
Bivalvia	<i>Macoma acolasta</i>	–	1	–	–	–	1	0.23	98.60
Crustacea	Cumacea	–	1	–	–	–	1	0.23	98.83
Crustacea	<i>Cancer productus</i>	–	–	1	–	–	1	0.23	99.07
Crustacea	<i>Rhepoxynius abronius</i>	–	–	–	1	–	1	0.23	99.30
Annelida	<i>Chaetozone setosa</i>	–	–	–	–	1	1	0.23	99.53
Platyhelminthes	Polycladida	–	–	–	–	1	1	0.23	99.77
Nematoda	Nematoda	–	–	–	–	1	1	0.23	100.00
Total Number of Individuals		184	203	166	95	119	767		
Total Number of Individuals Excluding <i>D. Excentricus</i>		101	103	71	65	89	429		

Table B.3 Replicate Sample Abundance by Taxa at Station B2 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	63	48	45	55	55	266	–	–
Mollusca	<i>Balcis rutila</i>	13	4	7	5	11	40	16.67	16.67
Annelida	<i>Magelona sacculata</i>	7	14	6	6	4	37	15.42	32.08
Crustacea	<i>Americhilidium shoemakeri</i>	3	4	3	4	17	31	12.92	45.00
Crustacea	<i>Rhepoxynius menziesi</i>	2	6	6	4	8	26	10.83	55.83
Annelida	<i>Apoprionospio pygmaea</i>	10	13	1	1	1	26	10.83	66.67
Nemertea	<i>Carinoma mutabilis</i>	2	7	4	2	1	16	6.67	73.33
Nemertea	<i>Cerebratulus californiensis</i>	–	6	2	2	4	14	5.83	79.17
Annelida	<i>Glycinde armigera</i>	1	1	1	–	3	6	2.50	81.67
Annelida	<i>Nephtys caecoides</i>	2	2	2	–	–	6	2.50	84.17
Bivalvia	<i>Tellina modesta</i>	2	–	2	1	1	6	2.50	86.67
Crustacea	<i>Rhepoxynius abronius</i>	2	–	–	3	–	5	2.08	88.75
Bivalvia	<i>Tellina bodegensis</i>	1	1	–	1	1	4	1.67	90.42
Annelida	<i>Armandia bioculata</i>	2	–	2	–	–	4	1.67	92.08
Crustacea	<i>Gibberosus myersi</i>	–	1	–	–	2	3	1.25	93.33

Table B.3 Replicate Sample Abundance by Taxa at Station B2 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Annelida	<i>Phyllodoce</i>	1	1	–	–	–	2	0.83	94.17
Annelida	<i>Sigalion spinosa</i>	1	–	–	–	1	2	0.83	95.00
Crustacea	Hippolytidae	2	–	–	–	–	2	0.83	95.83
Bivalvia	<i>Rochefortia tumida</i>	–	–	2	–	–	2	0.83	96.67
Annelida	<i>Lumbrineris</i>	1	–	–	–	–	1	0.42	97.08
Annelida	<i>Eumida</i>	1	–	–	–	–	1	0.42	97.50
Bivalvia	<i>Cooperella subdiaphana</i>	–	1	–	–	–	1	0.42	97.92
Annelida	<i>Lumbrineris californiensis</i>	–	1	–	–	–	1	0.42	98.33
Crustacea	<i>Cancer gracilis</i>	–	1	–	–	–	1	0.42	98.75
Mollusca	<i>Alvania</i>	–	–	1	–	–	1	0.42	99.17
Annelida	<i>Diopatra ornata</i>	–	–	1	–	–	1	0.42	99.58
Crustacea	<i>Blepharipoda occidentalis</i>	–	–	–	–	1	1	0.42	100.00
Total Number of Individuals		116	111	85	84	110	506		
Total Number of Individuals Excluding <i>D. Excentricus</i>		53	63	40	29	55	240		

Table B.4 Replicate Sample Abundance by Taxa at Station B3 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	58	29	50	36	47	220	–	–
Crustacea	<i>Rhepoxynius abronius</i>	2	–	2	2	–	6	2.46	2.46
Crustacea	<i>Rhepoxynius menziesi</i>	6	3	9	5	5	28	11.48	13.93
Annelida	<i>Scoloplos armiger</i>	–	2	1	1	1	5	2.05	15.98
Annelida	<i>Sigalion spinosa</i>	2	1	–	1	–	4	1.64	17.62
Crustacea	<i>Americhilidium shoemakeri</i>	3	1	3	2	6	15	6.15	23.77
Bivalvia	<i>Tellina bodegensis</i>	1	5	–	1	–	7	2.87	26.64
Bivalvia	<i>Tellina modesta</i>	2	3	4	1	–	10	4.10	30.74
Annelida	<i>Armandia bioculata</i>	–	2	–	4	3	9	3.69	34.43
Crustacea	<i>Gibberosus myersi</i>	–	–	–	–	1	1	0.41	34.84
Nemertea	<i>Carinoma mutabilis</i>	4	4	6	5	3	22	9.02	43.85
Nemertea	<i>Cerebratulus californiensis</i>	5	–	4	2	1	12	4.92	48.77
Bivalvia	<i>Chione</i>	–	–	–	1	1	2	0.82	49.59
Annelida	<i>Glycera convoluta</i>	–	1	–	–	–	1	0.41	50.00
Annelida	<i>Glycinde armigera</i>	1	2	2	1	–	6	2.46	52.46
Annelida	<i>Lumbrineris californiensis</i>	–	–	–	1	1	2	0.82	53.28
Annelida	<i>Lumbrineris</i>	–	–	–	1	–	1	0.41	53.69
Annelida	<i>Nephtys caecoides</i>	–	1	–	1	–	2	0.82	54.51
Crustacea	<i>Cancer gracilis</i>	1	–	–	–	–	1	0.41	54.92
Annelida	<i>Magelona sacculata</i>	4	5	7	6	3	25	10.25	65.16

Table B.4 Replicate Sample Abundance by Taxa at Station B3 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Platyhelminthes	Polycladida	–	–	–	1	–	1	0.41	65.57
Mollusca	Balcis rutila	9	17	15	14	19	74	30.33	95.90
Annelida	Apoprionospio pygmaea	–	3	1	1	2	7	2.87	98.77
Annelida	Tenonia priops	–	–	–	–	1	1	0.41	99.18
Crustacea	Cancer productus	–	–	–	–	1	1	0.41	99.59
Annelida	Syllides	–	–	1	–	–	1	0.41	100.00
Total Number of Individuals		98	79	105	87	95	464		
Total Number of Individuals Excluding <i>D. Excentricus</i>		40	50	55	51	48	244		

Table B.5 Replicate Sample Abundance by Taxa at Station B4 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	46	43	34	37	25	185	–	–
Mollusca	<i>Balcis rutila</i>	25	27	16	27	20	115	28.47	28.47
Annelida	<i>Magelona sacculata</i>	45	7	6	28	3	89	22.03	50.50
Crustacea	<i>Rhepoxynius menziesi</i>	6	9	9	6	–	30	7.43	57.92
Annelida	<i>Armandia bioculata</i>	17	–	1	7	1	26	6.44	64.36
Annelida	<i>Apoprionospio pygmaea</i>	19	2	–	3	1	25	6.19	70.54
Nemertea	<i>Carinoma mutabilis</i>	6	4	2	5	3	20	4.95	75.50
Annelida	<i>Glycinde armigera</i>	6	2	1	7	2	18	4.46	79.95
Nemertea	<i>Cerebratulus californiensis</i>	6	7	1	4	–	18	4.46	84.41
Annelida	<i>Sigalion spinosa</i>	4	2	–	2	–	8	1.98	86.39
Bivalvia	<i>Tellina bodegensis</i>	2	4	–	1	–	7	1.73	88.12
Crustacea	<i>Rhepoxynius abronius</i>	–	1	1	3	–	5	1.24	89.36
Annelida	<i>Scoloplos armiger</i>	–	1	1	3	–	5	1.24	90.59
Bivalvia	<i>Tellina modesta</i>	1	1	1	–	1	4	0.99	91.58
Crustacea	<i>Americhildium shoemakeri</i>	–	1	1	1	1	4	0.99	92.57
Bivalvia	<i>Cooperella subdiaphana</i>	–	2	–	1	1	4	0.99	93.56
Annelida	<i>Nephtys caecoides</i>	1	1	–	1	–	3	0.74	94.31
Annelida	<i>Gyptis brevipalpa</i>	1	1	–	–	–	2	0.50	94.80
Crustacea	Hippolytidae	2	–	–	–	–	2	0.50	95.30
Nemertea	<i>Tubulanus</i>	2	–	–	–	–	2	0.50	95.79
Annelida	<i>Syllides</i>	2	–	–	–	–	2	0.50	96.29
Crustacea	<i>Foxiphalus xiximeus</i>	–	1	–	1	–	2	0.50	96.78
Crustacea	<i>Gibberosus myersi</i>	1	–	–	–	–	1	0.25	97.03
Annelida	<i>Lumbrineris californiensis</i>	1	–	–	–	–	1	0.25	97.28
Annelida	<i>Tenonia priops</i>	1	–	–	–	–	1	0.25	97.52

Table B.5 Replicate Sample Abundance by Taxa at Station B4 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Vertebrata	<i>Chilara taylori</i>	1	–	–	–	–	1	0.25	97.77
Annelida	<i>Lumbrineris</i>	–	1	–	–	–	1	0.25	98.02
Annelida	<i>Phyllodoce</i>	–	1	–	–	–	1	0.25	98.27
Bivalvia	<i>Rochefortia tumida</i>	–	–	1	–	–	1	0.25	98.51
Crustacea	<i>Cancer gracilis</i>	–	–	1	–	–	1	0.25	98.76
Annelida	<i>Chaetozone setosa</i>	–	–	–	1	–	1	0.25	99.01
Annelida	<i>Diopatra ornata</i>	–	–	–	1	–	1	0.25	99.26
Annelida	<i>Glycera convoluta</i>	–	–	–	–	1	1	0.25	99.50
Nemertea	<i>Paranemertes sp. A</i>	–	–	–	–	1	1	0.25	99.75
Nemertea	<i>Lineus bilineatus</i>	–	–	–	–	1	1	0.25	100.00
Total Number of Individuals		195	118	76	139	61	589		
Total Number of Individuals Excluding <i>D. Excentricus</i>		149	75	42	102	36	404		

Table B.6 Replicate Sample Abundance by Taxa at Station B5 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	18	32	38	52	34	174	–	–
Mollusca	<i>Balcis rutila</i>	11	42	34	16	30	133	33.76	33.76
Annelida	<i>Magelona sacculata</i>	9	17	7	10	30	73	18.53	52.28
Crustacea	<i>Rhepoxynius menziesi</i>	7	15	23	9	7	61	15.48	67.77
Nemertea	<i>Carinoma mutabilis</i>	9	3	5	–	5	22	5.58	73.35
Annelida	<i>Apoprionospio pygmaea</i>	3	7	–	–	6	16	4.06	77.41
Bivalvia	<i>Tellina bodegensis</i>	3	2	3	2	1	11	2.79	80.20
Annelida	<i>Glycinde armigera</i>	–	4	3	3	1	11	2.79	82.99
Nemertea	<i>Cerebratulus californiensis</i>	4	2	3	–	1	10	2.54	85.53
Annelida	<i>Armandia bioculata</i>	–	6	–	–	4	10	2.54	88.07
Bivalvia	<i>Tellina modesta</i>	1	1	2	–	2	6	1.52	89.59
Crustacea	<i>Rhepoxynius abronius</i>	3	3	–	–	–	6	1.52	91.12
Annelida	<i>Sigalion spinosa</i>	2	1	–	–	–	3	0.76	91.88
Annelida	<i>Gyptis brevipalpa</i>	–	1	1	1	–	3	0.76	92.64
Annelida	<i>Chaetozone setosa</i>	–	1	–	–	2	3	0.76	93.40
Annelida	<i>Nephtys caecoides</i>	–	–	2	–	1	3	0.76	94.16
Nemertea	<i>Paranemertes sp. A</i>	–	1	1	–	–	2	0.51	94.67
Crustacea	<i>Cancer gracilis</i>	–	1	–	1	–	2	0.51	95.18
Annelida	<i>Phyllodoce</i>	–	–	1	–	1	2	0.51	95.69
Bivalvia	<i>Cooperella subdiaphana</i>	–	–	–	1	1	2	0.51	96.19
Annelida	<i>Glycera convoluta</i>	–	–	–	1	1	2	0.51	96.70
Bivalvia	<i>Siliqua lucida</i>	1	–	–	–	–	1	0.25	96.95
Echinodermata	<i>Leptosynapta</i>	1	–	–	–	–	1	0.25	97.21
Crustacea	<i>Lepidopa californica</i>	1	–	–	–	–	1	0.25	97.46
Annelida	<i>Harmothoe</i>	1	–	–	–	–	1	0.25	97.72

Table B.6 Replicate Sample Abundance by Taxa at Station B5 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Vertebrata	<i>Porichthys notatus</i>	1	–	–	–	–	1	0.25	97.97
Annelida	<i>Scoloplos armiger</i>	–	1	–	–	–	1	0.25	98.22
Annelida	<i>Lumbrineris</i>	–	1	–	–	–	1	0.25	98.48
Nemertea	<i>Tubulanus</i>	–	1	–	–	–	1	0.25	98.73
Nemertea	<i>Lineus bilineatus</i>	–	1	–	–	–	1	0.25	98.98
Crustacea	<i>Americhilidium shoemakeri</i>	–	–	1	–	–	1	0.25	99.24
Annelida	<i>Diopatra ornata</i>	–	–	1	–	–	1	0.25	99.49
Annelida	<i>Lumbrineris californiensis</i>	–	–	–	–	1	1	0.25	99.75
Bivalvia	<i>Rocheffortia tumida</i>	–	–	–	–	1	1	0.25	100.00
Total Number of Individuals		75	143	125	96	129	568		
Total Number of Individuals Excluding <i>D. Excentricus</i>		57	111	87	44	95	394		

Table B.7 Replicate Sample Abundance by Taxa at Station B6 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	20	32	23	54	32	161	–	–
Mollusca	<i>Balcis rutila</i>	87	47	20	24	32	210	42.86	42.86
Annelida	<i>Magelona sacculata</i>	8	13	6	17	9	53	10.82	53.67
Annelida	<i>Armandia bioculata</i>	43	1	–	–	–	44	8.98	62.65
Annelida	<i>Apoprionospio pygmaea</i>	26	4	4	–	5	39	7.96	70.61
Crustacea	<i>Rhepoxynius menziesi</i>	5	7	7	10	2	31	6.33	76.94
Nemertea	<i>Carinoma mutabilis</i>	3	3	3	4	4	17	3.47	80.41
Annelida	<i>Glycinde armigera</i>	1	2	4	2	4	13	2.65	83.06
Nemertea	<i>Cerebratulus californiensis</i>	1	3	1	2	2	9	1.84	84.90
Annelida	<i>Sigalion spinosa</i>	3	–	4	–	–	7	1.43	86.33
Bivalvia	<i>Tellina bodegensis</i>	1	–	3	1	1	6	1.22	87.55
Annelida	<i>Nephtys caecoides</i>	1	1	1	1	1	5	1.02	88.57
Nemertea	<i>Paranemertes sp. A</i>	1	1	–	3	–	5	1.02	89.59
Bivalvia	<i>Tellina modesta</i>	1	3	–	–	1	5	1.02	90.61
Crustacea	<i>Cancer productus</i>	5	–	–	–	–	5	1.02	91.63
Annelida	<i>Phyllodoce</i>	–	–	1	1	2	4	0.82	92.45
Annelida	<i>Scoloplos armiger</i>	1	1	1	–	–	3	0.61	93.06
Annelida	<i>Glycera convoluta</i>	1	–	1	–	1	3	0.61	93.67
Crustacea	<i>Americhilidium shoemakeri</i>	1	–	–	2	–	3	0.61	94.29
Nemertea	<i>Tubulanus</i>	3	–	–	–	–	3	0.61	94.90
Annelida	<i>Capitella capitata</i>	3	–	–	–	–	3	0.61	95.51
Bivalvia	<i>Siliqua lucida</i>	1	1	–	–	–	2	0.41	95.92
Annelida	<i>Eumida</i>	2	–	–	–	–	2	0.41	96.33
Crustacea	<i>Cancer gracilis</i>	–	1	–	1	–	2	0.41	96.73
Bivalvia	<i>Cooperella subdiaphana</i>	–	1	–	–	1	2	0.41	97.14

Table B.7 Replicate Sample Abundance by Taxa at Station B6 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Leptosynapta</i>	–	–	–	2	–	2	0.41	97.55
Crustacea	<i>Foxiphalus xiximeus</i>	1	–	–	–	–	1	0.20	97.76
Annelida	<i>Gyptis brevipalpa</i>	1	–	–	–	–	1	0.20	97.96
Crustacea	Aoridae	1	–	–	–	–	1	0.20	98.16
Annelida	<i>Prionospio cirrifera</i>	1	–	–	–	–	1	0.20	98.37
Annelida	<i>Diopatra ornata</i>	–	1	–	–	–	1	0.20	98.57
Crustacea	Hippolytidae	–	1	–	–	–	1	0.20	98.78
Cnidaria	Actiniaria	–	1	–	–	–	1	0.20	98.98
Annelida	<i>Pectinaria californiensis</i>	–	–	1	–	–	1	0.20	99.18
Crustacea	<i>Lepidopa californica</i>	–	–	–	1	–	1	0.20	99.39
Annelida	<i>Harmothoe</i>	–	–	–	1	–	1	0.20	99.59
Crustacea	<i>Rhepoxynius abronius</i>	–	–	–	–	1	1	0.20	99.80
Mollusca	<i>Alvania</i>	–	–	–	–	1	1	0.20	100.00
Total Number of Individuals		222	124	80	126	99	651		
Total Number of Individuals Excluding <i>D. Excentricus</i>		202	92	57	72	67	490		

Table B.8 Replicate Sample Abundance by Taxa at Station B7 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Echinodermata	<i>Dendraster excentricus</i>	42	63	28	53	52	238	–	–
Mollusca	<i>Balcis rutila</i>	22	18	9	33	15	97	28.28	28.28
Annelida	<i>Magelona sacculata</i>	24	20	2	6	3	55	16.03	44.31
Annelida	<i>Apoprionospio pygmaea</i>	16	4	11	2	–	33	9.62	53.94
Crustacea	<i>Rhepoxynius menziesi</i>	8	12	1	2	7	30	8.75	62.68
Annelida	<i>Armandia bioculata</i>	11	7	7	–	–	25	7.29	69.97
Annelida	<i>Sigalion spinosa</i>	4	3	3	–	1	11	3.21	73.18
Annelida	<i>Glycinde armigera</i>	7	4	2	–	–	13	3.79	76.97
Nemertea	<i>Carinoma mutabilis</i>	2	5	3	3	1	14	4.08	81.05
Bivalvia	<i>Tellina modesta</i>	2	2	–	3	–	7	2.04	83.09
Bivalvia	<i>Tellina bodegensis</i>	3	1	1	2	–	7	2.04	85.13
Annelida	<i>Nephtys caecoides</i>	1	2	1	1	–	5	1.46	86.59
Nemertea	<i>Cerebratulus californiensis</i>	3	1	–	3	–	7	2.04	88.63
Crustacea	<i>Cancer gracilis</i>	–	–	–	2	2	4	1.17	89.80
Annelida	<i>Capitella capitata</i>	2	3	–	–	–	5	1.46	91.25
Annelida	<i>Scoloplos armiger</i>	1	1	1	–	–	3	0.87	92.13
Annelida	<i>Lumbrineris californiensis</i>	1	–	1	–	1	3	0.87	93.00
Bivalvia	<i>Cooperella subdiaphana</i>	2	1	–	–	–	3	0.87	93.88
Crustacea	Hippolytidae	–	–	–	2	–	2	0.58	94.46
Nemertea	<i>Paranemertes sp. A</i>	–	1	–	–	1	2	0.58	95.04

Table B.8 Replicate Sample Abundance by Taxa at Station B7 during October 2016

Taxonomic Group	Taxon	Replicate					Total	Percent of Total	Cumulative Percent
		1	2	3	4	5			
Annelida	<i>Chaetozone setosa</i>	–	2	–	–	–	2	0.58	95.63
Annelida	<i>Glycera convoluta</i>	1	–	–	–	–	1	0.29	95.92
Peanut Worms	Sipuncula	1	–	–	–	–	1	0.29	96.21
Annelida	<i>Phyllodoce</i>	1	–	–	–	–	1	0.29	96.50
Annelida	<i>Ampharete labrops</i>	1	–	–	–	–	1	0.29	96.79
Bivalvia	<i>Siliqua lucida</i>	–	1	–	–	–	1	0.29	97.08
Crustacea	<i>Americhilidium shoemakeri</i>	–	1	–	–	–	1	0.29	97.38
Annelida	<i>Lumbrineris</i>	–	1	–	–	–	1	0.29	97.67
Echinodermata	<i>Amphiodia</i>	–	1	–	–	–	1	0.29	97.96
Annelida	<i>Syllides</i>	–	–	1	–	–	1	0.29	98.25
Mollusca	<i>Alvania</i>	–	–	–	1	–	1	0.29	98.54
Bivalvia	<i>Rochefortia tumida</i>	–	–	–	1	–	1	0.29	98.83
Annelida	<i>Nephtys cornuta franciscana</i>	–	–	–	–	1	1	0.29	99.13
Nemertea	<i>Tubulanus</i>	–	–	–	–	1	1	0.29	99.42
Nemertea	<i>Lineus bilineatus</i>	–	–	–	–	1	1	0.29	99.71
Annelida	<i>Spiophanes bombyx</i>	–	–	–	–	1	1	0.29	100.00
Total Number of Individuals		155	154	71	114	87	581		
Total Number of Individuals Excluding <i>D. Excentricus</i>		113	91	43	61	35	343		

Table B.9 Infaunal Indices in Sediment Samples Collected in October 2016

Parameter ^a		Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index (h)	Dominance (C')	Dominance (S _n)	Species Richness (d)	Evenness (J')	Infaunal Index (ITI)
Station B1	Replicate 1	184	183	16	11	1.64	1.53	0.30	3	2.88	0.59	53
	Replicate 2	203	199	17	12	1.48	1.37	0.35	2	3.02	0.52	46
	Replicate 3	166	166	17	10	1.48	1.35	0.38	2	3.13	0.52	80
	Replicate 4	95	93	15	6	2.06	1.85	0.19	4	3.09	0.76	71
	Replicate 5	119	116	19	6	2.19	1.98	0.17	5	3.79	0.74	56
Station B2	Replicate 1	116	113	15	8	1.68	1.51	0.34	3	2.96	0.62	67
	Replicate 2	111	110	15	7	1.94	1.76	0.23	5	2.98	0.72	72
	Replicate 3	85	84	14	6	1.79	1.57	0.31	4	2.93	0.68	72
	Replicate 4	84	84	11	8	1.38	1.22	0.45	3	2.26	0.58	83
	Replicate 5	110	110	14	8	1.73	1.56	0.29	3	2.77	0.65	87
Station B3	Replicate 1	98	98	13	8	1.60	1.43	0.37	4	2.62	0.62	89
	Replicate 2	79	79	15	5	2.07	1.83	0.20	5	3.20	0.77	65
	Replicate 3	105	104	12	9	1.78	1.62	0.27	4	2.37	0.72	85
	Replicate 4	87	84	17	5	2.02	1.77	0.23	5	3.61	0.71	65
	Replicate 5	95	94	14	7	1.70	1.52	0.30	3	2.86	0.65	64
Station B4	Replicate 1	195	191	19	10	2.20	2.05	0.15	5	3.43	0.75	56
	Replicate 2	118	116	18	6	2.04	1.84	0.21	5	3.58	0.71	83
	Replicate 3	76	76	14	5	1.75	1.54	0.27	3	3.00	0.66	81
	Replicate 4	139	139	19	7	2.21	2.02	0.16	5	3.65	0.75	65
	Replicate 5	61	61	13	5	1.68	1.45	0.28	3	2.92	0.65	56
Station B5	Replicate 1	75	73	14	5	2.26	2.00	0.13	6	3.03	0.86	82
	Replicate 2	143	141	19	7	2.14	1.96	0.17	4	3.64	0.73	71
	Replicate 3	125	124	14	9	1.88	1.72	0.21	3	2.70	0.71	91
	Replicate 4	96	96	10	10	1.47	1.33	0.34	3	1.97	0.64	82
	Replicate 5	129	128	17	8	2.00	1.83	0.19	4	3.30	0.71	66
Station B6	Replicate 1	222	216	23	9	1.96	1.82	0.23	4	4.09	0.63	35
	Replicate 2	124	123	18	7	1.92	1.74	0.23	4	3.53	0.67	74
	Replicate 3	80	79	14	6	2.10	1.86	0.17	5	2.98	0.79	78
	Replicate 4	126	122	13	9	1.72	1.58	0.26	3	2.50	0.67	79
	Replicate 5	99	96	14	7	1.82	1.64	0.24	3	2.85	0.69	72
Station B7	Replicate 1	155	153	19	8	2.29	2.11	0.14	5	3.58	0.78	58
	Replicate 2	154	151	19	8	2.06	1.89	0.22	5	3.59	0.70	62
	Replicate 3	71	71	14	5	1.98	1.73	0.21	4	3.05	0.75	49
	Replicate 4	114	113	13	9	1.60	1.45	0.31	2	2.54	0.62	72
	Replicate 5	87	86	12	7	1.38	1.22	0.41	2	2.47	0.56	88
Mean	Station B1	153	151	16.8	9.0	1.77	1.61	0.28	3.2	3.18	0.63	61
	Station B2	101	100	13.8	7.3	1.70	1.52	0.32	3.6	2.78	0.65	76
	Station B3	93	92	14.2	6.6	1.84	1.63	0.27	4.2	2.93	0.69	74
	Station B4	118	117	16.6	6.8	1.98	1.78	0.21	4.2	3.32	0.70	68
	Station B5	114	112	14.8	7.7	1.95	1.77	0.21	4.0	2.93	0.73	78
	Station B6	130	127	16.4	7.6	1.90	1.73	0.23	3.8	3.19	0.69	67
	Station B7	116	115	15.4	7.4	1.86	1.68	0.26	3.6	3.05	0.68	66
Standard Deviation	Station B1	45	45	1.5	2.7	0.34	0.29	0.10	1.3	0.35	0.12	14.0
	Station B2	15	15	1.6	0.7	0.21	0.20	0.08	0.9	0.30	0.05	8.6
	Station B3	10	10	1.9	1.6	0.20	0.17	0.07	0.8	0.49	0.06	12.3
	Station B4	53	52	2.9	2.1	0.25	0.28	0.06	1.1	0.34	0.05	13.3
	Station B5	28	27	3.4	1.7	0.31	0.27	0.08	1.2	0.64	0.08	9.9
	Station B6	55	53	4.2	1.7	0.14	0.12	0.03	0.8	0.62	0.06	18.4
	Station B7	38	37	3.4	1.4	0.37	0.35	0.10	1.5	0.54	0.09	15.0

Table B.9 Infaunal Indices in Sediment Samples Collected in October 2016

Parameter ^a		Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index (h)	Dominance (C')	Dominance (S _h)	Species Richness (d)	Evenness (J')	Infaunal Index (ITI)
Station Total	Station B1	767	757	32	24	1.85	1.78	0.28	3	4.68	0.53	60
	Station B2	506	501	23	22	1.86	1.78	0.31	5	3.54	0.59	74
	Station B3	464	459	22	21	1.94	1.85	0.27	4	3.43	0.63	73
	Station B4	589	583	31	19	2.23	2.15	0.17	5	4.71	0.65	64
	Station B5	568	562	29	19	2.13	2.04	0.19	4	4.42	0.63	76
	Station B6	651	636	30	21	2.14	2.07	0.19	5	4.49	0.63	56
	Station B7	581	574	29	20	2.11	2.02	0.22	5	4.41	0.63	61
Survey	Total	4126	4072	45	90	2.15	2.13	0.21	5	5.29	0.57	66

Table B.10 Infaunal Indices excluding *D. excentricus* in Sediment Samples Collected in October 2016

Parameter ^a		Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index (h)	Dominance (C')	Dominance (S _h)	Species Richness (d)	Evenness (J')	Infaunal Index (ITI)
Station B1	Replicate 1	101	100	15	6.67	1.75	1.56	0.31	3	3.04	0.65	53
	Replicate 2	103	99	16	6.19	1.58	1.39	0.40	3	3.26	0.57	46
	Replicate 3	71	71	16	4.44	1.86	1.60	0.29	5	3.52	0.67	80
	Replicate 4	65	63	14	4.50	2.12	1.84	0.18	6	3.14	0.80	71
	Replicate 5	89	86	18	4.78	2.19	1.93	0.18	5	3.82	0.76	56
Station B2	Replicate 1	53	50	14	3.57	2.25	1.91	0.14	7	3.32	0.85	67
	Replicate 2	63	62	14	4.43	2.23	1.94	0.14	6	3.15	0.84	72
	Replicate 3	40	39	13	3.00	2.36	1.96	0.11	7	3.28	0.92	72
	Replicate 4	29	29	10	2.90	2.13	1.74	0.13	5	2.67	0.92	83
	Replicate 5	55	55	13	4.23	2.06	1.78	0.17	5	2.99	0.80	87
Station B3	Replicate 1	40	40	12	3.33	2.26	1.90	0.12	6	2.98	0.91	89
	Replicate 2	50	50	14	3.57	2.24	1.90	0.16	7	3.32	0.85	65
	Replicate 3	55	54	11	4.91	2.10	1.83	0.15	5	2.51	0.88	85
	Replicate 4	51	48	16	3.00	2.34	1.96	0.14	6	3.87	0.84	65
	Replicate 5	48	47	13	3.62	2.02	1.70	0.21	5	3.12	0.79	64
Station B4	Replicate 1	149	145	18	8.06	2.17	1.99	0.17	5	3.42	0.75	56
	Replicate 2	75	73	17	4.29	2.20	1.91	0.18	6	3.73	0.78	83
	Replicate 3	42	42	13	3.23	1.92	1.60	0.22	4	3.21	0.75	81
	Replicate 4	102	102	18	5.67	2.22	1.99	0.17	6	3.68	0.77	65
	Replicate 5	36	36	12	3.00	1.70	1.36	0.33	4	3.07	0.68	56
Station B5	Replicate 1	57	55	13	4.23	2.26	1.95	0.13	6	2.99	0.88	82
	Replicate 2	111	109	18	6.06	2.08	1.86	0.20	5	3.62	0.72	71
	Replicate 3	87	86	13	6.62	1.82	1.63	0.24	4	2.69	0.71	91
	Replicate 4	44	44	9	4.89	1.70	1.46	0.24	3	2.11	0.77	82
	Replicate 5	95	94	16	5.88	1.94	1.73	0.22	4	3.30	0.70	66
Station B6	Replicate 1	202	196	22	8.91	1.82	1.68	0.27	3	3.98	0.59	35
	Replicate 2	92	91	17	5.35	1.82	1.60	0.30	4	3.55	0.64	74
	Replicate 3	57	56	13	4.31	2.11	1.82	0.18	6	2.98	0.82	78
	Replicate 4	72	68	12	5.67	1.86	1.64	0.22	3	2.61	0.75	79
	Replicate 5	67	64	13	4.92	1.78	1.54	0.29	4	2.89	0.69	72
Station B7	Replicate 1	113	111	18	6.17	2.34	2.12	0.13	6	3.61	0.81	58
	Replicate 2	91	88	18	4.89	2.37	2.10	0.13	6	3.80	0.82	62
	Replicate 3	43	43	13	3.31	2.15	1.81	0.15	5	3.19	0.84	49
	Replicate 4	61	60	12	5.00	1.71	1.47	0.33	4	2.69	0.69	72
	Replicate 5	35	34	11	3.09	1.79	1.46	0.25	4	2.84	0.75	88

Table B.10 Infaunal Indices excluding *D. excentricus* in Sediment Samples Collected in October 2016

Parameter ^a		Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index (h)	Dominance (C')	Dominance (S _h)	Species Richness (d)	Evenness (J')	Infaunal Index (ITI)
Mean	Station B1	86	84	15.8	5.3	1.90	1.66	0.27	4.4	3.36	0.69	61
	Station B2	48	47	12.8	3.6	2.21	1.87	0.14	6.0	3.08	0.87	76
	Station B3	49	48	13.2	3.7	2.19	1.86	0.16	5.8	3.16	0.85	74
	Station B4	81	80	15.6	4.9	2.04	1.77	0.21	5.0	3.42	0.75	68
	Station B5	79	78	13.8	5.5	1.96	1.73	0.21	4.4	2.94	0.76	78
	Station B6	98	95	15.4	5.8	1.88	1.65	0.25	4.0	3.20	0.70	67
	Station B7	69	67	14.4	4.5	2.07	1.79	0.20	5.0	3.23	0.78	66
Standard Deviation	Station B1	17	17	1.5	1.04	0.25	0.22	0.09	1.3	0.32	0.09	14.0
	Station B2	13	13	1.6	0.70	0.11	0.10	0.02	1.0	0.26	0.05	8.6
	Station B3	6	5	1.9	0.73	0.13	0.10	0.03	0.8	0.50	0.05	12.3
	Station B4	46	45	2.9	2.08	0.23	0.28	0.07	1.0	0.29	0.04	13.3
	Station B5	28	27	3.4	0.96	0.22	0.19	0.05	1.1	0.58	0.07	9.9
	Station B6	60	58	4.2	1.79	0.13	0.11	0.05	1.2	0.55	0.09	18.4
	Station B7	33	32	3.4	1.28	0.31	0.32	0.09	1.0	0.48	0.06	15.0
Station Total	Station B1	429	419	31	13.52	2.09	1.98	0.26	5	4.97	0.61	60
	Station B2	240	235	22	10.68	2.49	2.34	0.11	7	3.85	0.81	74
	Station B3	244	239	21	11.38	2.39	2.24	0.14	7	3.65	0.78	73
	Station B4	404	398	30	13.27	2.35	2.24	0.16	6	4.84	0.69	64
	Station B5	394	388	28	13.86	2.18	2.07	0.19	5	4.53	0.66	76
	Station B6	490	475	29	16.38	2.11	2.02	0.23	5	4.54	0.63	56
	Station B7	343	336	28	12.00	2.44	2.31	0.14	6	4.64	0.73	61
Survey	Total	2544	2490	44	56.59	2.43	2.39	0.16	7	5.50	0.64	66

SUSAN J. WILLIAMS

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Education

B.S.	Marine Biology, California State University, Long Beach
Certificate	Environmental Studies, California State University, Long Beach
M.S.	Biology, California State University, Long Beach

Credential

Instructor, California Community Colleges
Subjects: Biological Sciences, Marine Sciences, Ecology

Qualifications

Susan Williams is an environmental scientist with over thirty years experience in ecological monitoring and assessment. She has a strong background in natural history and an excellent working knowledge of such varied communities as coastal sage scrub, coastal dunes, wetlands, and marine benthos. Ms. Williams also has training in hazardous materials management, including courses and seminars in CEQA, environmental auditing, and the fate of toxins in the environment. She has an extensive background in biological survey techniques and laboratory methodology.

The postgraduate Environmental Studies Certificate provided interdisciplinary training in economics, public health issues, environmental toxicology, and assessing human impacts on terrestrial and marine environments.

Professional Employment

- 1989-present. Consultant and Contract Employee. Projects include Shoreline Recovery Study of the Exxon Valdez spill, terrestrial ecological survey of a Superfund site, and various benthic surveys, and environmental education.
- 1990-present. Interpretive Specialist, City of Ventura (Community Services). As part of the Interpretive Outreach Program, deliver classroom and field presentations on a variety of natural and cultural history topics; program development.
- 1993 - 1994. Park Ranger (Interpretive Division), Channel Islands National Park. Staff Visitor Center, present programs to school groups and general public, develop public programs (Tidepools, Island Ethnobotany).
- 1989 - 1994. Part-time Instructor, Oxnard College. Topics taught include Microbiology, Marine Biology, Biology, and Environmental Sciences.
- 1986 - 1989. Researcher, Battelle Ocean Sciences, Ventura. Laboratory supervisor, task leader for monitoring program, taxonomist, safety officer.
- 1984 - 1986. Research Scientist, Harbors Environmental Project, University of Southern California. Assessment of the marine environment of Marina Del Rey; monitoring offshore disposal of cannery waste.
- 1980 - 1986. Assistant Curator, Allan Hancock Foundation, University of Southern California. Responsible for maintaining museum collection of marine animals, processing loans, conducting independent research.

- 1979 - 1981. Research Associate, Applied Ecology Program, Institute for Marine and Coastal Studies. Investigated impacts of storms on sandy beaches, monitored benthic impacts of electrical generating stations, supervised laboratory.
- 1975 - 1978. Associate Research Scientist, Department of Biological Sciences, University of Southern California. As part of the Outer Continental Shelf Program, collected and analyzed benthic samples prior to offshore oil exploration.

Miscellaneous

Charter Member, Southern California Association of Marine Invertebrate Taxonomists

Participant, Boys & Girls Club of Ventura Career Fair

Judge, Ventura County Science Fair (2004-present). Assigned categories included Zoology, Botany, and Human Biology, junior division (grades 6-8)

Publications and Technical Reports

Ms. Williams has published 13 papers, ranging from taxonomic works and ecological analyses to articles on environmental issues for the general public.

Detailed list available upon request.

ROY K. KROPP

Senior Research Scientist
Pacific Northwest National Laboratory
Battelle Marine Sciences Laboratory
Sequim, Washington

Education

Ph.D.	Zoology, University of Maryland, 1988
M.S.	Biology, University of Guam, 1982
B.S.	Zoology, San Diego State University, 1978

Qualifications

Dr. Kropp is a specialist in benthic marine ecology, toxicology, and the systematics of crustaceans and mollusks with 21 years of experience. He has served as the principal investigator for or participated in marine environmental surveys in the tropical and boreal Pacific, off California, the Gulf of Mexico, along the Atlantic Coast of the United States, and in the Mediterranean. Currently, Dr. Kropp is the Senior Scientist for Benthic Biology for the Massachusetts Water Resources Authority Monitoring Program. Since matriculating to the Marine Sciences Laboratory from Battelle's Duxbury facility, Dr. Kropp has continued benthic ecological studies, recently completing studies of infaunal communities associated with dredged material disposal sites in Rhode Island Sound and Long Island Sound. He was the technical project manager for the preparation of the Final Environmental Impact Statement being prepared for the Providence River dredging project by the U.S. Army Corps of Engineers and is participating as a benthic ecologist on similar EISs for the designation of disposal sites in Rhode Island Sound and Long Island Sound. Dr. Kropp designed and directed a study to examine the effects of increased summer nutrient loads on nearshore seagrass beds in NW Sardinia by examining changes in plant phenology, motile epifaunal communities, and plant epiphyte loads. Dr. Kropp is a recognized authority on crustacean systematics. He was the crustacean taxonomist for the Sardinian study and served as the crustacean and mollusk taxonomist for studies conducted off California and New England. He was the crustacean taxonomist for the Gulf of Thailand and Sea of Okhotsk programs and has extensive experience in the systematics of Indo-Pacific crustaceans. Dr. Kropp is a Research Associate with the Smithsonian Institution and a Research Affiliate of the University of Guam Marine Laboratory. He has authored or coauthored many journal publications and technical reports.

Project Experience

- **Soft-Bottom Monitoring Study conducted for Massachusetts Water Resources Authority (MWRA).** Senior Scientist responsible for technical oversight of benthic monitoring tasks, including directing data analyses and report preparation for study of the response of the Boston Harbor benthos to the abatement of sludge discharge, and for the eventual diversion of effluent to Massachusetts Bay. Assisted in the design and performance of the rapid assessment technique used in the study. Co-principal Investigator of a study to determine the importance of the new outfall site as a recruitment area for juvenile lobsters.
- **Piscataqua River, New Hampshire 301(h) waiver Water Quality and Benthic Community Study.** Principal investigator for the study performed to fulfill requirements of a 301(h) waiver application. Designed the sampling scheme for the study, analyzed ecological and water-quality data, and prepared the interpretive report.
- **Rhode Island Sound and Long Island Sound Infaunal Community Analyses.** Co-Principal Investigator for two studies of infaunal communities in and near dredged material disposal sites for the U.S. Army Corps of Engineers. Performed benthic data analyses and coauthored the technical reports.
- **Effect of Water-Column Nutrients on Seagrass Epifaunal Communities in NW Sardinia, Italy.** Principle Investigator responsible for study design, staff training, study implementation, data analysis and interpretation, and preparation of interpretive report. Study examining the effects of increased summer nutrient loads (related to increased tourism) on nearshore seagrass beds by examining changes in plant phenology, motile epifaunal communities, and epiphyte loads.
- **Environmental Studies in the Gulf of Thailand, Phase I and II** Biology Task Leader for project that investigated the impacts of drilling activities on the benthos. Performed ecological data analysis, wrote biological characterization chapters of the resulting reports.

Roy K. Kropp (Continued)

Other Experience

Monitoring Studies.

Task Leader for a study to determine the abundance of ocean quahogs (*Arctica islandica*) at proposed dredged material disposal sites in Rhode Island Sound. Designed and directed the study. Analyzed quahog abundance and size-frequency data, and prepared an interpretive report.

Biology Task Leader for the Sediment Survey at the (New York) Mud Dump Site and Environs. Responsible for analysis and interpretation of infaunal data collected during 1994. Prepared sections on the benthic community for the draft final report. Contributed sections on the infaunal community and trophic relationships for the draft SEIS.

Task Leader. Statistical design of field sampling and data analysis, interpretation, and write-up for program involving lead shot remediation at Remington Gun Club site.

Biology Task Leader for Environmental Studies in the Sea of Okhotsk. Analyzed infaunal data collected from proposed drilling platform sites. Prepared an interpretive report on the infaunal communities in the study area.

Biology Task Leader for the Region VI (EPA) contaminated sediment study. Analyzed infaunal data from seven ocean dredged material disposal sites in the Gulf of Mexico. Prepared interpretive reports on the communities at each site and provided recommendations for improving the study design.

Scientist for the New York Bight Rapid Benthic Assessment project that concerned the selection of a new site for the disposal of dredge spoils. Conducted epibenthic trawls and sediment grab sampling. Analyzed trawl catches, fish stomach contents, and infauna collected in sediment grabs.

Assistant Program Manager for California OCS Phase II Monitoring Program (CAMP) that established baseline conditions in the Santa Maria Basin prior to the installation of a drilling platform. Principal Investigator studying the life histories of several key infaunal species. Conducted independent study on the incidence of repaired shell damage to gastropods and scaphopods in the study area.

Beach Replenishment Dredging Studies. Principal Investigator for three USACE sand borrow area studies. Responsible for design and ecological analysis for three projects establishing baseline conditions or investigating the effects of dredging in sand borrow areas at Broadkill Beach in Delaware Bay and off Absecon Inlet and Great Egg Harbor, New Jersey.

Toxicological Studies. Project Manager or Senior Scientist for toxicological testing of sediments from the Providence River, New Haven, and Boston Harbor/Weymouth Fore River. These tests were conducted from the New England Division, Army Corps of Engineers and involved marine species, including *Americamysis bahia* and *Ampelisca abdita*. Responsibilities included overall test planning and supervision, data analyses, and report preparation.

Project Manager for toxicological testing of sediments from Pistol Point Bar, CT. These tests were conducted from the New England Division, Army Corps of Engineers and involved freshwater species including *Hyalella azteca*, *Chironomus tentans*, and *Lumbriculus variegatus*.

Project Manager for toxicological testing of synthetic-based drilling muds from the Gulf of Mexico. These tests were conducted from the American Petroleum Institute and involved a marine species, *Leptocheirus plumulosus*.

Environmental Impact Statements. Project Manager for the preparation of the Final Environmental Impact Statement being prepared for the Providence River dredging project by the US Army Corps of Engineers. Coordinator for activities including response to public comments, re-analysis of fish trawl data to evaluate fishery value of potential disposal sites, directed the development of a statistically valid design of study on lobster populations at disposal sites, directing implementation of the study, designed a dredge survey to sample northern quahog populations in the area to be dredged and wrote the interpretive report, monitored project budgets and tracked deliverable preparation. Information will be used to assist the government in deciding the most appropriate disposal alternative.

Artificial Reef Studies. Benthic Infauna/Epifauna Task Leader for 5-year study for U.S. Army Corps of Engineers (USACE) and Environmental Protection Agency (EPA) Delaware Artificial Reef Study that investigated the feasibility of using artificial reef as compensation for shallow- and deep-water habitat loss. Performed ecological data analysis of epifaunal communities on the reefs and infaunal communities surrounding the reefs, wrote sections

Roy K. Kropp (Continued)

of the report. Designed portions of a study to examine the effects of artificial reef units on the infauna.

NOAA Hydrolab Aquanaut and field assistant for study of artificial reef colonization at Salt River Canyon, St. Croix, U.S. Virgin Islands.

Taxonomist. Crustacean taxonomist for benthic studies conducted in NW Sardinia, Italy.

Identified crustaceans and mollusks collected from Boston Harbor, Massachusetts Bay, and Cape Cod Bay for the MWRA Harbor and Outfall Monitoring Program.

Chief taxonomist for the Environmental Studies in the Gulf of Thailand, Phase I and II project. Identified the crustaceans collected during the study. Directed the establishment of a project-specific voucher collection and edited the documentation for the voucher collection.

Identified crustaceans and mollusks collected from off California for the CAMP program and the Morro Bay/Cayucos Sanitation District 301(h) monitoring program.

Specialist on benthic invertebrates for several baseline monitoring studies conducted in the western Pacific (Micronesia).

Professional Affiliations

Biological Society of Washington
Crustacean Society
Estuarine Research Federation
Pacific Science Association

Technical Reports

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APPENDIX C

Properties of Sediment Samples Collected during the October 2016 Offshore Survey

Table C.1 Sediment Grab-Sample Collection Record

Station	Sample	Date	Time ¹	Penetration Depth (cm)	Latitude	Longitude	Sediment Type	Sediment ² Color	Comment
1	Bio 1	1025/2016	10:45:23	9.0	35° 23.7432' N	120° 52.6785' W	Sand	Lt. Brown	<i>D. ornata</i> tube
	Bio 2	1025/2016	10:47:42	8.5	35° 23.7423' N	120° 52.6764' W	Sand	Lt. Brown	
	Bio 3	1025/2016	10:50:20	9.5	35° 23.7404' N	120° 52.6735' W	Sand	Lt. Brown	
	Bio 4	1025/2016	10:52:47	7.5	35° 23.7393' N	120° 52.6709' W	Sand	Lt. Brown	
	Bio 5	1025/2016	10:55:05	9.5	35° 23.7379' N	120° 52.6703' W	Sand	Lt. Brown	
	Chem 1	1025/2016	10:57:24	N/A	35° 23.7376' N	120° 52.6679' W	Sand	Lt. Brown	
	Chem 2	1025/2016	11:02:03	N/A	35° 23.7375' N	120° 52.6670' W	Sand	Lt. Brown	
	Chem 3	1025/2016	11:06:22	N/A	35° 23.7381' N	120° 52.6658' W	Sand	Lt. Brown	
2	Bio 1	1025/2016	10:13:39	9.5	35° 23.2712' N	120° 52.4965' W	Sand	Lt. Brown	
	Bio 2	1025/2016	10:19:31	8.5	35° 23.2716' N	120° 52.4973' W	Sand	Lt. Brown	
	Bio 3	1025/2016	10:21:53	10.0	35° 23.2719' N	120° 52.4956' W	Sand	Lt. Brown	
	Bio 4	1025/2016	10:24:20	8.0	35° 23.2709' N	120° 52.4935' W	Sand	Lt. Brown	
	Bio 5	1025/2016	10:26:43	9.5	35° 23.2712' N	120° 52.4932' W	Sand	Lt. Brown	
	Chem 1	1025/2016	10:09:10	N/A	35° 23.2697' N	120° 52.4914' W	Sand	Lt. Brown	
	Chem 2	1025/2016	10:16:05	N/A	35° 23.2711' N	120° 52.4956' W	Sand	Lt. Brown	
	Chem 3	1025/2016	10:29:11	N/A	35° 23.2707' N	120° 52.4927' W	Sand	Lt. Brown	
3	Bio 1	1025/2016	9:36:40	8.5	35° 23.2394' N	120° 52.5063' W	Sand	Lt. Brown	
	Bio 2	1025/2016	9:39:12	8.5	35° 23.2407' N	120° 52.5089' W	Sand	Lt. Brown	
	Bio 3	1025/2016	9:41:49	7.5	35° 23.2417' N	120° 52.5090' W	Sand	Lt. Brown	
	Bio 4	1025/2016	9:44:18	9.5	35° 23.2418' N	120° 52.5095' W	Sand	Lt. Brown	
	Bio 5	1025/2016	9:51:08	9.5	35° 23.2412' N	120° 52.5077' W	Sand	Lt. Brown	
	Chem 1	1025/2016	9:46:55	N/A	35° 23.2417' N	120° 52.5089' W	Sand	Lt. Brown	
	Chem 2	1025/2016	9:53:40	N/A	35° 23.2405' N	120° 52.5072' W	Sand	Lt. Brown	
	Chem 3	1025/2016	9:57:16	N/A	35° 23.2413' N	120° 52.5073' W	Sand	Lt. Brown	
4	Bio 1	1025/2016	9:09:59	8.0	35° 23.2134' N	120° 52.5120' W	Sand	Lt. Brown	<i>D. ornata</i> tube
	Bio 2	1025/2016	9:12:30	7.0	35° 23.2120' N	120° 52.5151' W	Sand	Lt. Brown	
	Bio 3	1025/2016	9:15:06	9.0	35° 23.2132' N	120° 52.5132' W	Sand	Lt. Brown	
	Bio 4	1025/2016	9:17:29	10.0	35° 23.2130' N	120° 52.5114' W	Sand	Lt. Brown	
	Bio 5	1025/2016	9:24:48	7.5	35° 23.2136' N	120° 52.5116' W	Sand	Lt. Brown	
	Chem 1	1025/2016	9:02:41	N/A	35° 23.2142' N	120° 52.5064' W	Sand	Lt. Brown	
	Chem 2	1025/2016	9:06:33	N/A	35° 23.2122' N	120° 52.5070' W	Sand	Lt. Brown	
	Chem 3	1025/2016	9:20:02	N/A	35° 23.2128' N	120° 52.5110' W	Sand	Lt. Brown	
5	Bio 1	1025/2016	8:23:14	9.0	35° 23.1847' N	120° 52.5012' W	Sand	Lt. Brown	<i>D. ornata</i> tube <i>D. ornata</i> tube
	Bio 2	1025/2016	8:25:52	10.5	35° 23.1829' N	120° 52.5009' W	Sand	Lt. Brown	
	Bio 3	1025/2016	8:28:38	8.0	35° 23.1834' N	120° 52.5021' W	Sand	Lt. Brown	
	Bio 4	1025/2016	8:31:09	9.0	35° 23.1793' N	120° 52.5068' W	Sand	Lt. Brown	
	Bio 5	1025/2016	8:33:58	8.5	35° 23.1772' N	120° 52.5091' W	Sand	Lt. Brown	
	Chem 1	1025/2016	8:36:47	N/A	35° 23.1840' N	120° 52.5085' W	Sand	Lt. Brown	
	Chem 2	1025/2016	8:41:21	N/A	35° 23.1860' N	120° 52.5084' W	Sand	Lt. Brown	
	Chem 3	1025/2016	8:45:18	N/A	35° 23.1851' N	120° 52.5089' W	Sand	Lt. Brown	

¹ Time when the sediment grab was collected (trip time of the grab)

² No odors emanating from the sediment samples were noted at any station

Table C.1 Sediment Grab-Sample Collection Record

Station	Sample	Date	¹ Time	Penetration Depth (cm)	Latitude	Longitude	Sediment Type	Sediment ² Color	Comment
6	Bio 1	1025/2016	7:53:16	9.0	35° 23.1761' N	120° 52.5085' W	Sand	Lt. Brown	
	Bio 2	1025/2016	7:56:23	7.5	35° 23.1749' N	120° 52.5096' W	Sand	Lt. Brown	
	Bio 3	1025/2016	7:59:03	9.0	35° 23.1755' N	120° 52.5088' W	Sand	Lt. Brown	
	Bio 4	1025/2016	8:01:38	9.0	35° 23.1764' N	120° 52.5063' W	Sand	Lt. Brown	
	Bio 5	1025/2016	8:04:23	7.0	35° 23.1756' N	120° 52.5057' W	Sand	Lt. Brown	
	Chem 1	1025/2016	7:48:59	N/A	35° 23.1751' N	120° 52.5093' W	Sand	Lt. Brown	
	Chem 2	1025/2016	8:06:58	N/A	35° 23.1764' N	120° 52.5065' W	Sand	Lt. Brown	
	Chem 3	1025/2016	8:11:22	N/A	35° 23.1767' N	120° 52.5063' W	Sand	Lt. Brown	
7	Bio 1	1025/2016	7:08:52	10.0	35° 23.1289' N	120° 52.5102' W	Sand	Lt. Brown	
	Bio 2	1025/2016	7:12:16	9.5	35° 23.1283' N	120° 52.5109' W	Sand	Lt. Brown	
	Bio 3	1025/2016	7:15:37	7.0	35° 23.1286' N	120° 52.5115' W	Sand	Lt. Brown	
	Bio 4	1025/2016	7:18:58	10.5	35° 23.1311' N	120° 52.5089' W	Sand	Lt. Brown	
	Bio 5	1025/2016	7:21:32	8.0	35° 23.1302' N	120° 52.5107' W	Sand	Lt. Brown	
	Chem 1	1025/2016	7:24:21	N/A	35° 23.1314' N	120° 52.5070' W	Sand	Lt. Brown	
	Chem 2	1025/2016	7:28:36	N/A	35° 23.1319' N	120° 52.5067' W	Sand	Lt. Brown	
	Chem 3	1025/2016	7:32:11	N/A	35° 23.1321' N	120° 52.5075' W	Sand	Lt. Brown	

Table C.2 Grain-Size Class Distribution

Phi	-2 to -1	-1 to 0	0 to 1	1 to 2	2 to 3	3 to 4	4 to 9	>9
Size (µm)	2000	1000	500	250	125	62.5	62.5 to 2	<2
Class	Gravel	Vry Crs Snd	Crs Sand	Med Snd	Fine Snd	Vry Fine Snd	Silt	Clay
Station 1	0.107%	0.075%	0.091%	0.610%	70.050%	27.328%	1.236%	0.503%
Station 1(d)	0.242%	0.089%	0.169%	0.887%	73.041%	23.897%	1.057%	0.618%
Station 2	0.162%	0.050%	0.073%	0.894%	74.448%	22.887%	0.992%	0.493%
Station 3	0.475%	0.060%	0.107%	0.565%	72.304%	24.960%	0.940%	0.591%
Station 4	0.154%	0.086%	0.202%	1.218%	74.285%	22.938%	0.609%	0.508%
Station 4(d)	0.137%	0.087%	0.162%	1.117%	73.693%	23.491%	0.777%	0.536%
Station 5	0.332%	0.114%	0.129%	0.621%	73.819%	23.605%	0.872%	0.508%
Station 5(d)	0.064%	0.123%	0.215%	0.711%	72.670%	25.137%	0.646%	0.433%
Station 6	0.078%	0.076%	0.149%	0.814%	72.808%	24.895%	0.723%	0.458%
Station 7	0.001%	0.073%	0.104%	0.550%	76.822%	21.020%	0.919%	0.511%

Table C.3 Grain-Size Weight Distribution

Phi Size (µm) Class	Dry Weight (grams)			Percentage Summary		
	<-1 to 4 2000 to 62.5	>4 <62.5	Total	-2 to -1 2000	-1 to 4 1000 to 62.5	4 to >9 <62.5
	Coarse	Fine		Gravel	Sand	Mud
Station 1	114.462	2.026	116.488	0.107%	98.153%	1.739%
Station 1(d)	112.717	1.920	114.637	0.242%	98.084%	1.675%
Station 2	111.241	1.677	112.918	0.162%	98.353%	1.485%
Station 3	108.985	1.694	110.679	0.475%	97.994%	1.531%
Station 4	110.384	1.247	111.631	0.154%	98.729%	1.117%
Station 4(d)	111.726	1.487	113.213	0.137%	98.550%	1.313%
Station 5	108.553	1.519	110.072	0.332%	98.288%	1.380%
Station 5(d)	110.216	1.203	111.419	0.064%	98.856%	1.080%
Station 6	114.745	1.372	116.117	0.078%	98.741%	1.182%
Station 7	112.598	1.634	114.232	0.001%	98.569%	1.430%

Table C.4 Grain-Size Statistics

Station	Percent Coarser than Listed Diameter (µm)								
	Mean (φ)	σ (φ)	Skewness (φ)	Mean (µm)	5%	16%	50%	84%	95%
B1	2.85	0.391	0.22	139	203	177	143	106	79
B1(d)	2.80	0.392	0.21	144	210	182	147	110	81
B2	2.79	0.378	0.21	145	209	182	148	112	83
B3	2.82	0.386	0.21	142	207	180	146	109	81
B4	2.78	0.380	0.17	146	214	185	149	113	85
B4(d)	2.79	0.385	0.19	145	212	184	148	111	83
B5	2.80	0.376	0.20	144	208	181	147	111	83
B5(d)	2.81	0.372	0.18	143	206	180	146	110	84
B6	2.81	0.375	0.19	143	207	180	146	110	84
B7	2.79	0.352	0.22	145	203	179	149	115	85

AVAIL: Oct - last week
NOV
DEC
OR EARLY SPRING

Morro Bay/Cayucos Wastewater Treatment Plant
Outfall/Diffuser Annual Report

Name of Discharger: MBCSD
NPDES Permit Number: _____

Name of Diver/Inspector: Craig Porter / Porter Diving -
Firm & Address: 15730 Morro rd

Telephone Number: (805) 466 - 0566

Outfall Length: 5160 Feet Depth of Water: 50' Feet

Design of Diffuser (physical dimensions, shape, include sketch): _____

Number of Ports: 34 Number Ports Open: 28 Diffuser Level? 2' to 4' 6"

End of Diffuser Open? _____ Closed? X

Outfall Marked by Bouys? Yes: X No: _____ Type: SPAR

Diffusers Operating as Designed? Yes: X No: _____

Breaks in Outfall Line Noted? Yes: _____ No: X if yes, describe

Describe Problems Noted: NONE _____

Corrected Action Needed: NO _____

Physical Description of Environment Around Outfall (i.e. fish noted, debris, sediment, etc.):

Small fish, Sand Dollars, Crabs, Anemones

Direction of Plume in Relationship to Shoreline: North, North west

Pictures Taken? Yes: X video No: _____ (Include copies if taken)

[Signature]
Signature of Diver/Inspection

9-13-16
Date

Signature of Diver/Inspection

Date

**Table E.1 Summary of Analytical Methods Used for
Physicochemical Determinations of Sediment**

Parameter	Method	Description	Reference
Biochemical Oxygen Demand 5-Day	5210B	Dissolved Oxygen Probe	6
Total Kjeldahl Nitrogen (TKN)	351.2	Block Digestion/Colorimetry	1
Total Volatile Solids	160.4	Dry ashing/Gravimetry	1
Moisture	160.3	Evaporation/Gravimetry	1
Sediment Grain Size	MRS, 1998	Sieve/Pipette without salt	2,3,4
Aluminum	6010	ICP	5
Arsenic	7060	GFAAS	5
Cadmium	6010	ICP	5
Chromium	6010	ICP	5
Copper	6010	ICP	5
Iron	6010	ICP	5
Lead	7421	CVAAS	5
Mercury	7471	CVAAS	5
Nickel	6010	ICP	5
Silver	272.2	GFAAS	1
Zinc	6010	ICP	5
Oil and Grease	1664	Extraction/Gravimetry	7

CVAAS = Cold Vapor Atomic Absorption Spectrometry

ICP/MS = Inductively Coupled Plasma/Mass Spectrometry

GC = Gas Chromatography

GC/MS = Gas Chromatography/Mass Spectrometry

GFAAS = Graphite Furnace Atomic Absorption Spectrometry

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